Juvenile trout survival and movement during the summer low flow abstraction period in the Lindis River, Central Otago

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Summary

Survival and movement of juvenile brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) were investigated during low flow conditions in the Lindis River, Central Otago. High levels of abstraction from this river result in drought-like events on an annual basis. Over two years, during the summer low flow period >1000 juvenile trout were marked with passive integrated transponder (PIT) tags. Their movements were tracked using mobile PIT tag antenna. To detect emigration and losses to an irrigation channel stationary PIT tag antenna were also positioned on the main-stem river and on the channel. After PIT tagging the study area was subjected to extreme low flow conditions in both years. Mark-recapture model analysis indicated that only a small proportion of the sample population survived the low flow events; 0.34 (95 % CI: 0.21-0.50) in year one and 0.29 (95% CI: 0.18-0.39) in year two. Water temperatures in flowing water remained within salmonid tolerance levels. Low survival rates were primarily attributed to high levels of predation and the lowest survival probabilities were associated with reduced cover for juvenile salmonids. Fish that did die from heat stress in disconnected and drying pools were immediately removed by scavengers, indicating that any dead fish observed during prolonged low flow conditions are likely to represent a small proportion of the total mortality experienced. Potential fish movement from stressful habitat during the both years was constrained by drying reaches, and an irrigation intake structure. Some sample fish displayed considerable movement along the remaining fragmented river corridor, presumably in an attempt to find refuge habitat.

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Chapter 1: General Introduction

Summer drought can significantly reduce stream flow and place river ecosystems and fish populations under stress (Magoulick and Kobza 2003, Bond et al. 2008). The impact of naturally occurring drought events is compounded by excessive water abstraction (Leprieur et al. 2006, Bond et al. 2008, Lange et al. 2014). Abstraction for irrigation is the most common consumptive use of freshwater (Döll and Siebert 2002) and globally demand is increasing due to agricultural intensification (Poff et al. 2003). The impacts of excessive abstraction and flow alteration are becoming increasingly apparent. These include dry river beds, prolonged low flows (Postel 2000) and, in some cases mass fish kills, thought to be related to high temperatures and low oxygen levels (Levy 2003). In New Zealand many rivers and streams are also under pressure due to increasing demand for abstraction and irrigation development (Dewson et al. 2007, Kienzle and Schmidt 2008) and there is intense debate among resource managers, irrigators and environmental interest groups about how to protect river ecosystems and meet demands for irrigation (Poff et al. 2003).

Understanding the ecological consequences of water abstraction is essential for protecting river ecosystems and the fisheries they sustain (Milner et al. 2012, Mims and Olden 2013). Studies have shown that flow diversion can interrupt the longitudinal connectivity of rivers and fish migration routes (Greenberg and Calles 2010) and that low flows reduce both the area and quality of available habitat, which may impact survival as well as have sub-lethal effects for fish (Hayes et al. 2010, Jonsson and Jonsson 2011). However, relatively few studies have quantified the actual effects of flow reduction on fish survival and movement in the wild (Riley et al. 2009) or assessed the impact of excessive abstraction at the population level.

1.1 Impacts of low flow events on stream dwelling juvenile trout

Summer drought and low flow conditions can have negative impacts on juvenile salmonid populations in small streams (Elliott et al. 1997, Jonsson and Jonsson 2011). Trout populations in wide and shallow streams with little cover experience the greatest impacts and those in confined channels with overhanging vegetation and undercut banks the least (see review in Hayes and Young 2001). Effects may be both direct and indirect (Berger and Gresswell 2009) as a result of changes to both the quality and amount of available habitat (see review in Hayes and Young 2001, Berger and Gresswell 2009). Prolonged low flow events can increase water temperatures and decrease oxygen levels, resulting in reduced trout growth rates and weight loss (Elliott 1993, 2000, Harvey et al. 2005, Quinn 2011). In extreme events flow reduction can result in fish strandings and direct mortality due to thermal and oxygen stress (Elliott 2000, Hayes and Young 2001, Jonsson and Jonsson 2011). As flows decrease fine sediment deposition and algal proliferation can increase, which in turn reduces invertebrate drift and turnover of aquatic food resources (see review in Hayes and Young 2001, Hakala and Hartman 2004, Matthaei et al. 2010). Reduced flows and habitat can also increase competition for territory and food, and result in a density dependent reduction of apparent survival rates (Richard et al. 2013a).

Summer drought flows can reduce cover for juvenile salmonids in small streams by decreasing the availability of habitat such as riffles and undercut banks (Hakala and Hartman 2004), and increase predation risk in some instances (Heggenes and Borgstrøm 1988, Boss and Richardson 2002, Riley et al. 2009). Fish confined and concentrated in small pool habitat during low flows may become more vulnerable to avian predation (Richard et al. 2013a). However, predation events can be unpredictable and difficult to quantify. For example, the serendipitous discovery of vulnerable prey in a drying stream by a predator may result in sudden and severe mortality (Boss and Richardson 2002). Many studies have failed to determine a relationship between cover reduction and predation during low flow events (Berger and Gresswell 2009). Boss and Richardson (2002) found the addition of cover reduced juvenile salmonid mortality by 50 % in one stream, but had no apparent effect in another stream. The decrease in mortality was thought to be due to a reduction in predation attacks when cover was present (Boss and Richardson 2002). Predation resulted in 4 - 60 % mortality of salmonid fry in various Massachusetts streams with decreasing riffle habitat associated with increased predation (Henderson and Letcher 2003). However, the relationship between riffle cover and predation was not consistent (Henderson and Letcher 2003).

1.2 Juvenile trout movement during low flow events

Juvenile trout can undertake small scale movements and migration events in response to low flows (Elliott 2000, Landergren 2004). Small scale movement is often seen as fish moving from areas of high temperature stress, such as shallow runs and riffles, to deeper and cooler pools (Elliott 2000, see review in Hay 2004). Instinctual behaviour and environmental factors are thought to trigger juvenile trout migration from their natal stream (Elliott 1994, Jonsson and Jonsson 2011). The most important environmental factors are considered to be temperature and flow at the time fish are ready to move (Elliott 1994, Jonsson and Jonsson 2011). Migration of juvenile salmonids from their natal rearing streams is usually downstream (Jonsson and Jonsson 2011). Upstream movement in response to summer low flows or increasing temperature has been documented (Gowan and Fausch 1996, Kahler et al. 2001, Davey and Kelly 2007). Severe summer drought and reduced flows can block juvenile migration attempts due to loss of longitudinal connection (Elliott 2000). The loss of longitudinal river connectivity can fragment river habitat and fish populations and disrupt critical migratory corridors for juvenile salmonids (Greenberg and Calles 2010).

Brown trout populations display remarkable life history variations (Kristensen 2006, Jonsson and Jonsson 2011). In New Zealand migration of juvenile trout has been recorded in all months of the year (see review in Jellyman and Bonnett 1992). A study of juvenile brown trout in the Rainy River near Nelson found migration occurred mostly during autumn and spring freshes, and to a lesser extent during spring low flow periods. Migration rates during summer low flows were estimated to be minimal (Holmes et al. 2013). However, in the lower Silver Stream located near Dunedin juvenile brown trout cohorts that did not experience high discharge events were thought to leave their natal rearing stream during summer low flow conditions (Kristensen and Closs 2008). The out-migrating fry were a similar size to some that stayed, indicating that competition for habitat could not solely explain movement, and an instinctual behaviour may have triggered migration (Kristensen 2006). The variation in reported juvenile trout migration patterns highlights that more research is required to better understand this behaviour.

1.3 Study aims

While small scale movements of salmonids in response to flow reduction have been well documented, more research is needed to better understand the relationship between low flows and long distance movements (Campbell and Scott 1984, Huntingford et al. 1999, see review in Hay 2004). Similarly, studies that have quantified juvenile trout survival during low flows are scarce, making comparisons with previous work difficult (Elliott et al. 1997, Riley et al. 2009). Most New Zealand research on the impacts of flow reduction on salmonids has focused on theoretical relationships between flow and physical habitat (Hayes et al. 2010). A notable exception is the work of Holmes et al. (2013) that employed Passive Integrated Transponder (PIT) technology to track individual juvenile brown trout in the Rainy River, a headwater tributary of the Motueka River, which experiences unaltered hydrology (no abstraction or diversion). Holmes et al. (2013) found that migration comprised 60 % and mortality 29 % of losses over a nine month period; with little movement or mortality predicted during the summer low flow period.

To investigate whether similar patterns occur in a highly altered system, I aimed to assess juvenile trout survival and movement in the Lindis River where abstraction results in extreme flow reduction for an extended period each summer. The overall objectives of the study were to: i) quantify juvenile salmonid survival during the low flow abstraction period and, ii) investigate if juvenile salmonids moved from stressful habitat during the low flow abstraction period.

1.4 Study area

The Lindis River is located in the semi-arid Central Otago region of New Zealand (Figure 1.1), which is currently undergoing development and intensification of irrigated agriculture (Kienzle and Schmidt 2008) . In this region excessive levels of abstraction can compound the impact of natural drought events on rivers (Leprieur et al. 2006). In addition, historical mining privileges, which permit the over-allocation of many rivers and are now used for irrigation, will expire in 2021 and must be replaced by Resource Management Act (1991) consents. These resource consents will be subject to minimum or residual flows set by the Otago Regional Council (ORC) following consultation with irrigators and stakeholder groups. While historic over-allocation of water resources presents some difficult management challenges, it also presents an opportunity to study the effects of extreme flow reduction on fish populations and inform the minimum flow setting process.

The Lindis River is a third order tributary of the Upper Clutha River located in Central Otago, New Zealand (Figure 2.1). It is approximately 55 km long and drains a catchment area of 1055 km^2. The lower Lindis Valley is one of the driest regions in New Zealand with an average annual rainfall of approximately 500 mm, and as little as 300 mm some years. In this region droughts are a natural climatic feature but their hydrological impacts are exacerbated by abstraction for irrigation (Leprieur et al. 2006). The estimated naturalized (without abstraction) mean annual low flow (MALF) of the lower Lindis is 1.860 m^3/s. However, after abstraction for irrigation the actual MALF is 0.177 m^3/s (ORC 2008, Horrell 2014). Abstraction results in the complete drying of approximately 10 km of lower river reaches in most summers. The surface flow reduction is compounded by groundwater losses in affected reaches (Jellyman and Bonnett 1992, ORC 2008). Substratum comprises mostly gravels (2-64 mm) and small cobbles (64 – 256 mm), with silt, sand and boulders present. Land cover in the headwaters is dominated by native tussock and low intensity grazed grassland. More intensively managed pasture, crop production and some viticulture occurs on the lower river flats. The Lindis River is considered to be an important spawning and nursery stream contributing recruitment to the Upper Clutha River and Lake Dunstan trout fisheries (Jellyman and Bonnett 1992, Turner 1994, ORC 2008). Young of the year (0+) and yearling (1+) juvenile brown and rainbow trout are the most common age classes present. Brown trout are most common with rainbows representing <10 % of the population (Jellyman and Bonnett 1992). The river also sustains a population of adult brown trout in the perennial reaches (ORC 2008). Adult rainbow trout are rarely seen during the angling season but do utilise the river and tributaries for spawning (Jellyman and Bonnett 1992).

Figure 1.1 Lindis River landscape 15 March 2014, note areas of intensive irrigation in a naturally semi-arid catchment

1.5 Thesis structure

This thesis is structured as four chapters, beginning with this introduction (Chapter 1). Chapter Two presents the methodology and results of a mark-recapture investigation undertaken to determine the fate of juvenile trout during low flow conditions. Chapter Three presents an analysis of trout movements during the low flow conditions. Chapters Two and Three are written in the format of individual scientific papers. Because of this style, and the use of a common mark-recapture dataset for survival and movement analysis, there is some repetition across chapters. Overall conclusions and recommendations are presented in Chapter Four.

Chapter 2: Juvenile trout survival during summer low flow events in the Lindis River

2.1 Abstract

To investigate the survival of juvenile brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) during summer low flow conditions in the Lindis River, Central Otago >1000 juvenile trout (age 0+ and 1+) were marked with Passive Integrated Transponder (PIT) tags, and tracked using mobile PIT tag antennas over two years. To detect emigration and losses to an irrigation channel, stationary PIT tag antennas were positioned on the main-stem river and on the channel. After PIT tagging, the study area was subjected to extreme low flow conditions due to high levels of water abstraction in both years. Flows reduced to approximately 25% of Mean Annual Low Flow (MALF) or less for 75 days in year one, and approximately 25 % of MALF or less for the entire 59 day study period in year two. Mark-recapture model analysis indicated that the low flow event resulted in very low survival of the sample population, 0.34 (95% CI: 0.21-0.50) in year one and 0.29 (95% CI: 0.18-0.39) in year two. Water temperatures and dissolved oxygen levels in flowing water remained within salmonid tolerance levels. Low survival rates were primarily attributed to high levels of predation, and the lowest survival probabilities were associated with reduced cover for juvenile salmonids. Fish that died from heat stress in disconnected and drying pools were immediately removed by scavengers. These results show that predictions of trout population response to water abstraction based on habitat modeling alone may be overly simplistic by failing to account for ecosystem response to low flows, such as increased vulnerability to predation.

2.2 Introduction

Density dependent factors such as competition for territory, and density independent factors such as flow variability, interact in a complex relationship that regulates juvenile trout populations (Armstrong et al. 2003, Milner et al. 2003, Jonsson and Jonsson 2011, Richard et al. 2013a). The highest rates of survival are generally associated with moderate flows, as opposed to flood or drought events (Lobón-Cerviá 2009). Stream flow during the early emergent life stage is a major determinant of overall cohort success (Lobón-Cerviá 2009). Approximately 80-90 % of salmonid eggs may be expected to survive and produce free swimming alevins (Jonsson and Jonsson 2011, Quinn 2011). This is provided spawning gravel has minimal fine material content which can smother developing eggs (Quinn 2011) and flooding does not occur (Hayes 1995b).

After emergence from gravels the young alevins enter a period of high mortality known as the critical period which can last for approximately 30-70 days (Elliott 1994, Jonsson and Jonsson 2011). During the critical period the majority of alevins rarely feed, and drift downstream and die, while the remainder compete for and establish feeding territories (Elliott 1993, Elliott 1994). At this stage the young alevins are highly vulnerable to predation (Jonsson and Jonsson 2011). Mortality levels during the critical period can be as high as 80-90 % (Elliott 1993, Elliott 1994, Jonsson and Jonsson 2011). In rearing streams with high juvenile densities, mortality rates are thought to be density-dependent (Elliott 1993, Elliott 1994, Jonsson and Jonsson 2011). Post-critical period mortality is largely governed by density-independent factors (Elliott 1994, Milner et al. 2003, Jonsson and Jonsson 2011). Generally post-critical period juveniles would be expected to experience a period of lower mortality (Allan 1951, Elliott 1993, Jonsson and Jonsson 2011). Exceptions to this more stable pattern can include high levels of mortality (Milner et al. 2003) as a result of severe summer droughts (Elliott et al. 1997) and major flood events (Hayes et al. 2010, Holmes et al. 2013). For anadromous salmonids high mortality rates also occur when the young move to sea and experience major changes in diet and habitat (Jonsson and Jonsson 2011, Quinn 2011). Other factors that can influence juvenile survival rates are parasites, disease and the abundance of predators (Elliott 1993, Quinn 2011). Predation rates can depend on the density, size and behaviour of the prey, and the predator species, as well as the presence of disease (Jonsson and Jonsson 2011).

There are very few studies that have estimated survival rates of juvenile trout impacted by drought and flow reduction (Elliott et al. 1997). One notable exception is an investigation of juvenile sea-run brown trout in a small English stream (Elliott et al. 1997). During severe drought years the densities of juveniles was reduced to 15 - 68 % of expected values. Lower survival combined with poor growth rates was shown to reduce the breeding potential of one affected year class (Elliott et al. 1997). A mark-recapture study of juvenile brook trout in small headwater streams in North America estimated a 67 % reduction of 0+ fish during a drought (Hakala and Hartman 2004). Survival estimates of juvenile cutthroat trout during summer low flow in two small North American streams varied between 40-100 %. In one stream the addition of cover reduced mortality by 50 %, probably because the cover reduced predation risk (Boss and Richardson 2002).

Few New Zealand studies have investigated the impacts of low flows on juvenile trout populations, and fewer still have been able to differentiate losses between migration and mortality. Jellyman and Bonnett (1992) undertook extensive electrofishing studies throughout the Lindis River catchment and extrapolated their results to estimate that 59 % of all juvenile recruitment could be lost during low flow events, with losses to unscreened irrigation races thought to be considerable. Kristensen (2006) documented the loss of juvenile trout in the lower Silver Stream during summer months that was thought to be attributed to migration, as opposed to in-stream mortality. Hayes et al. (2010) conducted a long term electrofishing investigation of the Rainy River juvenile trout population which indicated that the population was not significantly affected by natural flow reductions during the early summer-autumn period.

However, a key information gap in the aforementioned studies is that they were unable to partition losses to mortality or out-migration. Holmes et al. (2013) subsequently employed Passive Integrated Transponder (PIT) technology to identify and track individual fish, allowing the differentiation between migration and mortality in the Rainy River. This research indicated that spring and autumn migration formed 60 % of fish losses, and mortality which was largely caused by flooding accounted for 29 %, with 11% of the fish predicted to stay in the stream until the following year. Fish movement and mortality during the summer period was estimated to be relatively low (Holmes et al. 2013). My study aimed to compare these findings from a stream which experiences natural flow reduction with an assessment of survival in the Lindis River in Central Otago that experiences severe levels of flow reduction due to abstraction. The objectives were to i) determine juvenile trout survival during the low flow period and, ii) to investigate causes of mortality.

2.3 Methods

2.3.1 Study area

The Lindis River is subject to significant abstraction and experiences severe low flows on an annual basis (Jellyman and Bonnett 1992). To assess juvenile trout survival and movement across habitats which experience varying severity of flow reduction, the study area was categorised into three stream segments (Figure 2.1). These were based on observed differences in surface water connection due to variation in ground water losses (Jellyman and Bonnett 1992, ORC 2008). The higher stress segment experiences significant losses of surface water to ground (Jellyman and Bonnett 1992, ORC 2008) and approximately 50 % of this segment dries up completely during the summer abstraction period. Approximately 10 % of the intermediate stress segment riverbed dries up as a result of an irrigation channel diversion arm that diverts the surface water and dewaters the riverbed immediately downstream. In the lower stress segment, surface flows are significantly reduced but surface water connection is maintained. The irrigation channel (Figure 2.1) is a priority take and upstream abstractors must deliver a consistent amount of water to this point (Jellyman and Bonnett 1992).

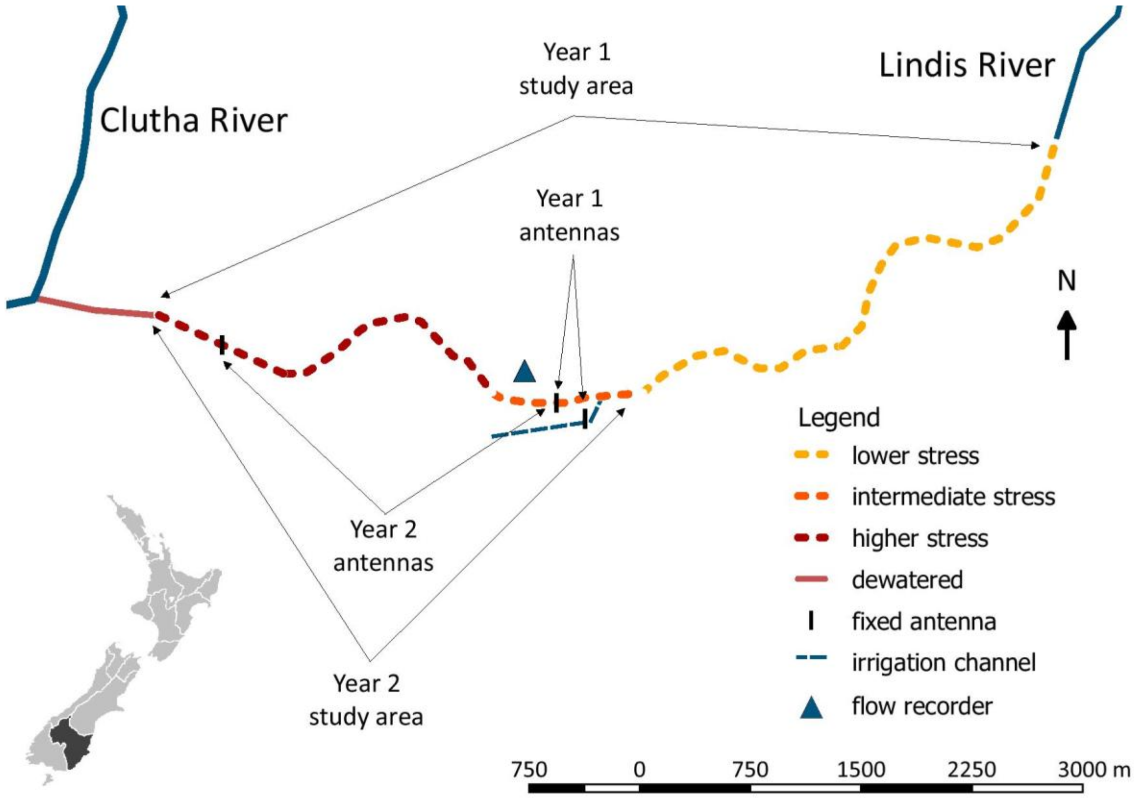

Figure 2.1 Study area showing location of higher, intermediate and lower stress stream segments, and fixed antenna positions, and study area during year one and year two.

Juvenile trout were captured, PIT tagged, and released into the stream segments. Survival was then monitored using fixed and mobile antenna over two summer-autumn low flow periods (year one and year two). In year one the study period extended from 21 January to 29 April 2014. To determine survival rates and movement patterns across a large spatial scale this investigation included approximately 9 km of riverbed comprising the higher, intermediate and lower stress segments. The total sample size of tagged fish was 622. In year two the spatial scale was reduced to the higher and intermediate segments only (approximately 3 km of riverbed). This allowed more intensive monitoring where the lowest survival rates had been identified in year one. The study period extended from 7 January to 13 March 2015. To investigate the cause of the low survival rates, additional monitoring included increased mobile antenna surveys, physical habitat investigations and the use of stationary cameras as described later. The total sample size of tagged trout in year two was 394. This included 99 fish sourced from a nearby tributary stream and released into the study area to increase sample size. These additional fish were also used to test PIT tag detection efficiency, as described later.

2.3.2 Passive integrated transponder tagging

Fish were captured for tagging by electrofishing using two Kainga EFM300 backpack electrofishing machines (NIWA Instruments, Christchurch NZ). A single electrofishing pass of each reach was conducted to minimise potential adverse effects that have been associated with electric fishing such as increased stress and forced movement (Nordwall 1999, Beaumont et al. 2000, Snyder 2003). Both electrofishing machines fished downstream in parallel, to pole nets that were used to retain fish. Captured trout were transferred to flow-through holding tubs so that the water remained cool and oxygenated and no additional stress was experienced prior to tagging. Prior to tagging, fish were anesthetized in a 0.2 ml/L solution of AQUI-S (a clove oil based fish anesthetic) so that they could be handled in a controlled and humane manner. The fork length (FL, nearest mm) of each fish was then measured to determine age class, and appropriate tag size.

In order to study age 0+ (young of year) and 1+ (yearling) juvenile trout in a manner that maximised probability of detection and minimised potential tag related stresses, two different tag sizes (12 mm and 23 mm) were used in age 0+ and 1+ fish respectively (Cucherousset et al.

2005, Zydlewski et al. 2006, Teixeira and Cortes 2007, Burnett et al. 2012). To minimise potential tagging stress, conservative limits for fish length were set regarding tag size (Richard et al. 2013b). Trout ≥60 - 130 mm FL were considered most likely to be age 0+ fish, based on previous studies of the trout population in the Lindis River (Jellyman 1990), and were tagged with half-duplex 12 mm PIT tags (12 mm x 2.15 mm in dimension, Oregon RFID, Portland, Oregon, U.S.A). A modified 12-gauge hypodermic needle was used to insert 12 mm tags into the fish body cavity anterior to the pectoral fin (Richard et al., 2013b, Acolas et al., 2007). Trout ≥130 mm were considered most likely to be 1+ (Jellyman 1990) and were tagged with 23 mm half-duplex PIT tags (23 mm x 3.65 mm in dimension, Oregon RFID, Portland, Oregon, U.S.A). To insert 23 mm tags, a scalpel was used to make a 4 mm incision along the mid-ventral line, anterior to the pelvic girdle. The tag was then inserted by hand and gently massaged forward into the fish's body cavity (Holmes et al., 2013). All tagged fish were adipose fin clipped to allow visual identification of marked fish. To minimize the risk of infection, all surgical equipment including tags was sterilized in 70 % ethanol prior to use. Following tagging fish recovered in a flow-through tub, and normal swimming behaviour was restored, before being released to calm water within their reach of capture.

2.3.3 Monitoring of sample fish

2.3.3.1 Fixed antenna

To determine if fish were attempting to migrate during the low flow period, two pass-through, stream width radio frequency identification (RFID) antennas were deployed each year (Figure 2.1). Each antenna was connected to a PIT tag reader (supplied by Oregon RFID) that was powered by 12 volt batteries and solar panels. In year one, the antennas were installed on the mainstem of river and in a nearby irrigation channel (Figure 2.1). This was to detect mainstem movement of fish, and to determine if fish were being entrained into the irrigation channel and lost from the system. In year two the second antenna was placed in mainstem river, within the higher stress segment (Figure 2.1). Fixed antennas were operational prior to tagging, and until the end of the study period each year.

2.3.3.2 Mobile antenna

To monitor fish that did not move past fixed antenna, fish were located throughout the study area with two types of mobile antenna. Each hand-held antenna was connected to a mobile backpack PIT tag reader unit. So that the large study area could be efficiently surveyed, initial detection was achieved using two large (3000 mm x 450 mm) rectangular antennas (Holmes et al. 2013) operated in tandem, sweeping downstream to systematically cover all areas (Cucherousset et al. 2010). The antenna design enabled access to water beneath debris and undercut banks. Coverage was increased in deeper areas by submerging the antenna. Two smaller, easy to maneuver hoop antennas (500 mm in diameter), were employed along shallow margins. Three complete mobile surveys of the study area were conducted in year one, and six in year two.

During both years fish movement out of the study area was restricted by areas of dry riverbed. In year one, the riverbed downstream of the study area dried during the tagging and release period. Fish could not move upstream of a section of dry riverbed 3 km above the study area. To confirm that sample fish did not move out of the study area a mobile antenna survey started at this point. Two subsequent mobile antenna surveys starting 700 m upstream of the study area were also conducted. In year two the riverbed downstream of the study area was dry throughout the entire study period and an irrigation diversion arm, which diverts almost all surface flow into an irrigation channel, prevented upstream passage of fish out of the study area. To confirm no sample fish had moved upstream of the study area, a mobile antenna survey from 2.3 km upstream of the diversion arm was conducted.

To determine the status of fish (alive or dead) each PIT tag detection event was checked by rescanning the detection area until the live fish was seen or moved and could no longer be detected. If the fish did not move a smaller wand-style antenna with a read range of 100 mm was used to locate the exact position of the PIT tag. The surrounding substrate was then moved by hand until the status of the fish (live or dead) could be confirmed or the tag was located. If a loose tag was detected the fish was assumed to be dead and decomposed (Holmes et al. 2013) because tag retention is generally very high when fish of a suitable size are marked (Richard et al. 2013b). Once the status of the tag detection was confirmed, the time and position was recorded using a hand held GPS unit (with approximately 25 m accuracy). This data was then later matched to the data on the mobile reader to give the time and position of each individual fish (or tag) detected.

2.3.3.3 Detection efficiency

To determine the detection efficiency of the mobile antenna for live tagged fish, a sample of 99 trout were captured, tagged and transferred from a nearby tributary stream into a 100 m stop netted reach located within the higher stress segment. This reach contained pool, run and riffle habitat typical of the study area. Fish were allowed to settle for one hour before a mobile antenna survey was conducted in the reach. An additional three survey passes were made within the stop netted reach with a rest period of 30 minutes between each pass to allow any disturbed fish to settle.

To determine the detection efficiency of mobile readers for loose PIT tags on the riverbed, 50 tags (45 x 12 mm and 5 x 23 mm, reflecting the approximate ratio of tag sizes used in the study) were randomly distributed in each of two 80 m reaches. One reach was located within the higher stress segment and one in the intermediate stress segment. Six mobile antenna surveys were conducted within the higher stress detection reach, and three within the intermediate detection reach, over a two week period, by operators who were unaware of the tag locations (blind test).

2.3.4 Investigating predation and scavenging

In year two stationary cameras with motion sensors (Spy Point model BF-6) were used to investigate predation activity and to determine whether predators were removing sample trout from the study area. Over three 24 hour periods two cameras were mounted overlooking two residual pools to monitor for predators where 30 and 21 sample fish respectively, were found stranded. Mobile antenna surveys were then conducted to monitor fish survival until the pools dried entirely.

To determine if fish that perished were being scavenged and if scavengers were removing PIT tags from the study area, 14 recently dead and 2 dying brown trout of various age classes (65 mm – 520 mm) were recovered from an almost dry pool in the lower reach of the stress segment. The dying fish were dispatched humanely. Once PIT tagged all fish were returned to the residual pool where a camera was set to monitor the site for 24 hours.

2.3.5 Assessment of physical habitat

Broad scale habitat assessments (Holmes et al. 2012) were conducted in year two in the segments where the lowest survival rates had been identified in year one (higher and intermediate stress segments). The reduced study area allowed more intensive monitoring to investigate the causes of the low survival rates. In each segment six 120 m reaches that contained representative pool, run and riffle habitat were selected and further divided into 20 m sub-reaches so that accurate measurement of habitat components could be made (Holmes et al. 2012). Within each 20 m reach meso-habitat types (e.g. pool, riffle, run), depths, wetted width, and potential juvenile trout cover such as woody debris, undercut banks, overhanging vegetation, surface turbulence, macrophyte cover (Hayes 1988a, Boss and Richardson 2002) and stream bed characteristics (e.g. substrate type, algal cover) were assessed. Each physical feature, such as the amount of undercut bank and overhanging vegetation, was expressed as a percentage of area surveyed.

2.3.5.1 Dissolved oxygen and temperature

To investigate if oxygen stress was likely to be a factor affecting mortality of sample fish a dissolved oxygen meter (YSI ProODO) was used to examine oxygen levels in pools that contained moving water over three 24 hour periods in season two. These readings were taken at mid-afternoon and before daylight when diurnal differences in dissolved oxygen may be expected to be the most extreme due to aquatic plant photosynthesis and respiration respectively (Wilcock et al. 1995). There were two sample sites located within the higher stress segment and one within the intermediate stress segment. To determine if high water temperature may be causing mortality of sample fish, instream temperature loggers (HOBO Water Temp Pro v2) were deployed within the higher and intermediate stress segments. These remained in position for the study period.

2.3.5.2 Flow Data

To measure the extent and duration of the low flow period, flow data was collected at Otago Regional Council monitoring station located within the intermediate stress segment (Figure 2.1).

2.3.6 Data analysis

2.3.6.1 Survival modeling

To estimate survival of fish during the low flow period in relation to time, stream segment, fish size and species, a Cormack-Jolly-Seber (CJS) model was used to conduct a mark-recapture analysis (Holmes et al. 2013) for both study years. For this analysis, each tag encounter was treated as a recapture event. Mobile and fixed antenna detections were combined to create an encounter history for each sample fish (Table 2.1). The study period for each year was divided into survey periods based around complete mobile survey efforts (Table 2.2 and Table 2.3). For each fish a code 1 was assigned if it was detected and confirmed to be alive during a survey period and a code 0 if it was not (Table 2.1). If a loose tag had been found the individual was assumed to be dead and decomposed (Holmes et al. 2013) and it was assigned a 0 as well. Multiple detections of the same fish within a survey period were treated as one encounter for that interval. All detections from the remote antenna were categorized as live fish (code 1), as fixed readers only detect tags that are orientated close to 90° it is unlikely antennas would detect a dead tagged fish tumbling in the current (Holmes et al. 2013).

Table 2.1 Example of the capture history of 6 individuals used for Cormack-Jolly-Seber models.

Encounter History	Stream Segment	Species	Size
1100	Intermediate stress	brown	large
1000	Intermediate stress	brown	large
1110	Higher stress	brown	small
1110	Higher stress	rainbow	small
1000	Lower stress	brown	small
1000	Lower stress	rainbow	small

Variables included in the CJS model included time, defined by survey period (Table 2.2 and Table 2.3), stream segment where fish were released, the stream segment where fish were detected, and fish size, defined as 0+/small (< 130 mm) or 1+/large (≥ 130 mm). Each survey period included at least one complete mobile survey of the study area. Unequal time intervals i.e. 30, 20 and 49 days, were used for year one. Four equal time periods of two week intervals were used for year two.

Table 2.2 Survey periods for Cormack-Jolly-Seber model year one.

Survey Period	Dates	Number of fish/detections
Release	21/01/2014 - 5/02/2014	622
1	7/02/2014 - 20/02/2014	133
2	25/02/2014 - 11/03/2014	151
3	12/03/2014 - 29/04/2014	114

Table 2.3 Survey periods for Cormack-Jolly-Seber model year two.

Survey Period	Dates	Number of fish/detections
Release	07/01/2015 - 13/01/2015	394
1	14/01/2015 - 27/01/2015	394
2	28/01/2015 - 12/02/2015	254
3	13/02/2015 - 20/02/2015	70
4	20/02/2015 - 06/03/2015	34

2.3.6.2 Habitat data analysis

To identify the most important variables for further analysis, a two-sample Kolmogorov-Smirnov test was conducted to determine which habitat variables (n = 25) were significantly different between the higher and intermediate stress segments. This test also indentified variables that were not significantly different and therefore unlikely to be related to differences in survival rates between the stream segments. A $P < 0.1$ criteria was set when assessing the physical habitat between the stream segments, reflecting the limited number of reaches sampled in each segment (n = 6), and to minimize the chance of excluding any variables that may be important (Effenberger et al. 2006). The variables indentified as significant (n = 9) were further explored from a multivariate perspective to determine which best explained differences between the segments (Allyon et al. 2009). A correlation-based principal component analysis (PCA) was performed. Analysis of similarities (ANOSIM) was performed using normalized Euclidean distances to determine if significantly different habitat types were present in the two stream segments.

2.4 Results

2.4.1 Capture and tagging

The capture and tagging period extended from 21 January to 5 February 2014 in year one, and from 7 to 13 January 2015 in year two. The details of the fish sampled, and the sample locations are summarised in Table 2.4. The average size of fish < 130 mm was 76 mm (± 1SD 12 mm), and for fish ≥130 mm was 175 mm (± 1SD 28 mm).

Table 2.4 Summary of year one and year two sample fish releases.

		Year One	Year Two
Stream segments investigated		Higher stress Intermediate stress Lower stress	Higher stress Intermediate stress
Fish sampled and released		622	394
Capture and tagging reaches by segment			
	Higher stress	2	14
	Intermediate stress	2	2
	Lower stress	18	
Sample fish released into each segment			
	Higher stress	86	310
	Intermediate stress	48	84
	Lower stress	488	
Age			
	0+ (young of the year)	82	21
	1+ (yearling)	540	373
Species			
	brown trout	556	394
	rainbow trout	66	

2.4.2 Passive integrated transponder encounters

Over both years there were a total of 1485 detection events, from 565 tags. The majority (1052) were recorded during mobile antenna surveys. All of these were confirmed to be from live fish except for 29 loose PIT tags that were found on the bed of the river and assumed to be from dead fish.

In year one the irrigation channel antenna recorded eight detections from seven tags, and the antenna located within the intermediate stress segment recorded 31 detections from 15 tags. In year two the antenna located within the higher stress segment recorded 67 detection events from 39 tags. The antenna located within the intermediate stress segment recorded 327 detection events from 100 tags.

2.4.2.1 Fixed Antenna Detection Efficiency

For an unknown reason (possibly electromagnetic disturbance) the detection range for the fixed antenna was limited to approximately 250 mm for 12 mm tags and could not be improved. The limited detection range of the fixed antenna meant that the mainstem antenna performance was compromised at flows greater than approximately 2.5 m^3/s, as fish could potentially move close to the stream's surface and pass over the antenna outside of the detection range. Flows exceeded 2.5 m^3/s at the end of year one (Figure 2.2), and did not exceed 2.5 m^3/s during year two (Figure 2.3). The irrigation channel antenna remained effective during year one as flows within the channel were confined to no more than 0.250 m^3/s, meaning that any passing PIT tagged fish would remain within the detection range.

2.4.2.2 Mobile Antenna Detection Efficiency

The mobile antenna had maximum detection ranges of 350 - 400 mm for 12 mm PIT tags and 550-700 mm for 23 mm PIT tags. Some variation in exact detection range was evident, possibly because of electromagnetic disruption at different sites along the river (Zydlewski et al. 2006, Connolly et al. 2008) and antenna angle in relation to the tag orientation (Hill et al. 2006). The average detection efficiency of the 10 tests conducted in the higher stress segment was 56 %, and of the 3 tests conducted in the intermediate stress segment it was 82 %. The difference may have been due to electromagnetic disturbance from sources such as power lines which can affect PIT tag performance (Zydlewski et al. 2006, Connolly et al. 2008) and were located approximately 50 m from the detection trials in the higher stress segment. The overall average of the 13 detection tests was 62.9 %. This figure was used to adjust final survival estimates generated by the CJS model. It is considered to be a conservative estimate of detection efficiency as most tests were conducted relatively close to suspected sources of electromagnetic disturbance.

2.4.2.3 Fish containment

During both years fish movement out of the study area was constrained by areas of dry riverbed. In year one the riverbed downstream of the study area dried during the tagging and release period. Fish could not move upstream of a dry reach 3 km above the study area. A mobile antenna survey started at this point and found no sample fish more the 200 m upstream from the study area. Two subsequent mobile antenna surveys starting 700 m upstream of the study area also found no sample fish more 200 m from the study area.

In year two the riverbed downstream of the study area was dry throughout the entire study period and an irrigation diversion arm, which diverts almost all surface flow into an irrigation channel, prevented upstream passage of fish out of the study area. To confirm no sample fish had moved upstream of the study area a mobile antenna survey from 2.3 km upstream of the diversion arm was conducted and recorded no tag detections. Two subsequent surveys starting 200 m upstream of the diversion arm also found no sample fish.

2.4.3 Flow observations

As spring progressed into summer and the height of the abstraction season in year one, river flows at the recorder decreased to very low levels (approximately 25% of MALF or less) and flat lined for much of the summer and autumn period (approximately 75 days in total) (Figure 2.2). Daily average flows during the study period varied between 0.25 m^3/s and a fresh which reached approximately 4 m^3/s and marked the end of the low flow period. All monitoring equipment (stationary antenna) was removed during a following fresh which reached 6.5 m^3/s.

As summer progressed into the height of the abstraction season in year two, river flows at the recorder in the intermediate stress segment decreased to very low levels (approximately 25% of MALF or less) and flat lined for the entire study period (59 days) (Figure 2.3). Daily average flows during the study period varied between 0.23 and 0.42 m^3/s.

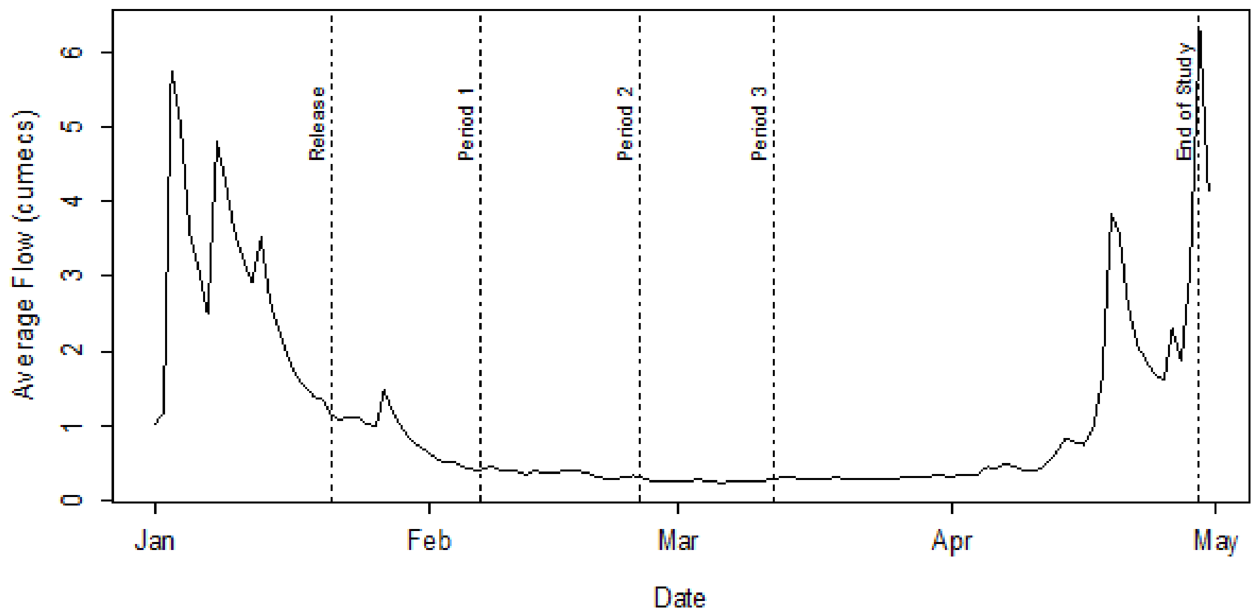

Figure 2.2 Year one study period, daily average flows (solid line) at the flow recorder located within the intermediate stress segment. The vertical dotted lines mark the start of each survey period.

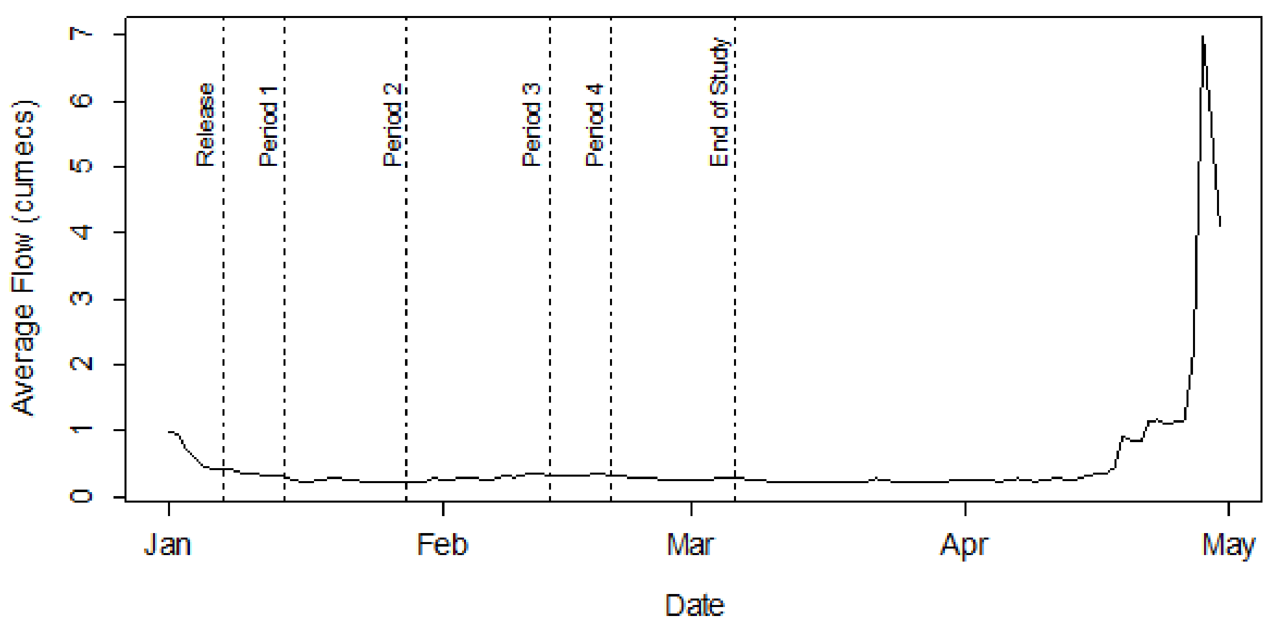

Figure 2.3 Year two study period, daily average flows (solid line) at the flow recorder located within the intermediate stress segment showing start and finish of study. The vertical dotted lines mark the start of each survey period.

2.4.4 Survival model

The CJS model of best fit in year one indicated that survival was most dependent on time (survey period) and release habitat (stream segment). All variables were initially included in the analysis. However, preliminary analysis showed that species had an insignificant effect on survival and this variable was excluded from subsequent analysis. In terms of survival, both time and release segment produced the best model of fit, indicating that survival depends on the habitat (stream segment) where fish were released and the habitat (stream segment) in which they were recorded during the survey periods.

The survival probabilities of sample fish for year one, and year two are shown in Figure 2.4 and Figure 2.5, respectively. Fish released into lower stress segments experienced higher survival probability than fish released into the higher and intermediate stress segments in both years. In year one, the lowest survival rate occurred in the first survey period, but in year two the lowest survival probabilities were experienced in the last two survey intervals. The overall survival estimates, after adjusting for detection efficiency (0.629), for the year one study period (98 days) were 0.34 (95% CI: 0.21–0.50), and 0.29 (95% CI: 0.18–0.39) during the year two study period (59 days). In both years an estimated >60 % of the sample population did not survive the first six weeks of the low flow event

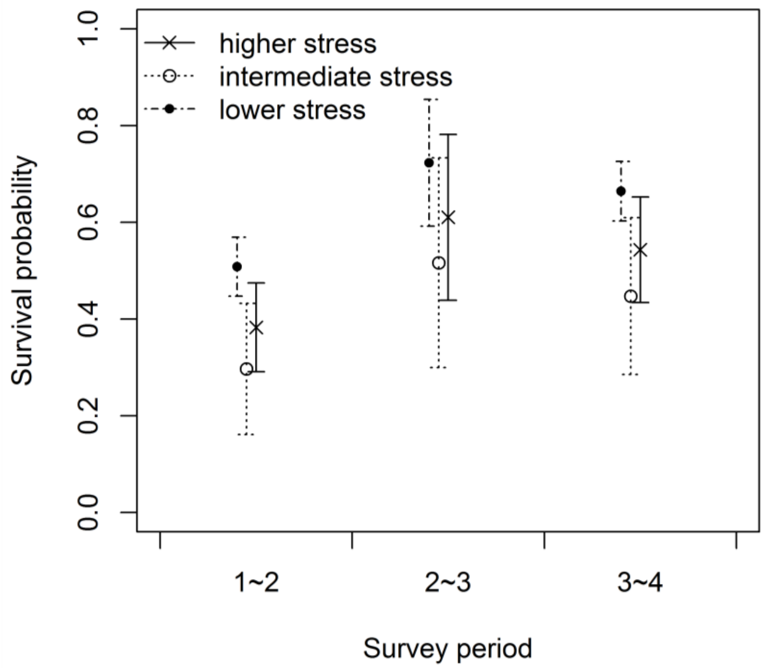

Figure 2.4 Survival probability as predicted by the Cormack-Jolly-Seber model for fish found in the intermediate, lower and higher stress habitats, over each survey period in year one

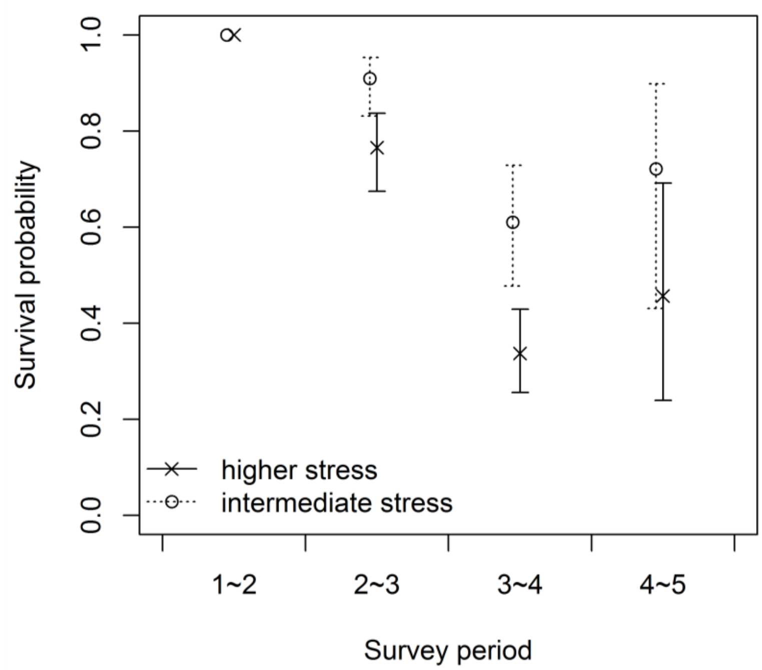

Figure 2.5 Survival probability as predicted by the Cormack-Jolly-Seber model for fish found in the intermediate and higher stress habitats, over each survey intervals in year two.

2.4.5 Causes of low survival

2.4.5.1 Predation and Scavenging Investigations

Twenty of the twenty nine loose tags found on the bed of the river over both years were under black shag perches, indicating that the fish into which these tags had been placed had been predated.

Of 30 PIT sample fish found stranded in a residual pool on 16 and 17 January of year two, 21 were redetected after two days (19 January) and just 10 of these were detected after another 8 days on 27 January. On the afternoon of 29 January the pool dried entirely and only one PIT tag was recovered. Of the 21 PIT tagged fish found stranded in another nearby residual pool on 19 January, just 9 were redetected on 27 January. On 29 January this pool also dried entirely and no PIT tags could be found. Trail cameras overlooking these residual pools revealed white faced herons feeding on live fish (Figure 2.6), as well as ferrets, rats and harrier hawks scavenging for dead fish. Large black shags and wild cats were also observed targeting this area during mobile antenna surveys.

Of the 16 dead trout that were PIT tagged and placed in a residual pool on 16 January, no fish or PIT tags could be found by the following morning. Motion cameras overlooking this residual pool detected rats, ferrets and white faced herons. A ferret was observed removing the largest trout (52cm long). By morning no trace of fish or tags could be found.

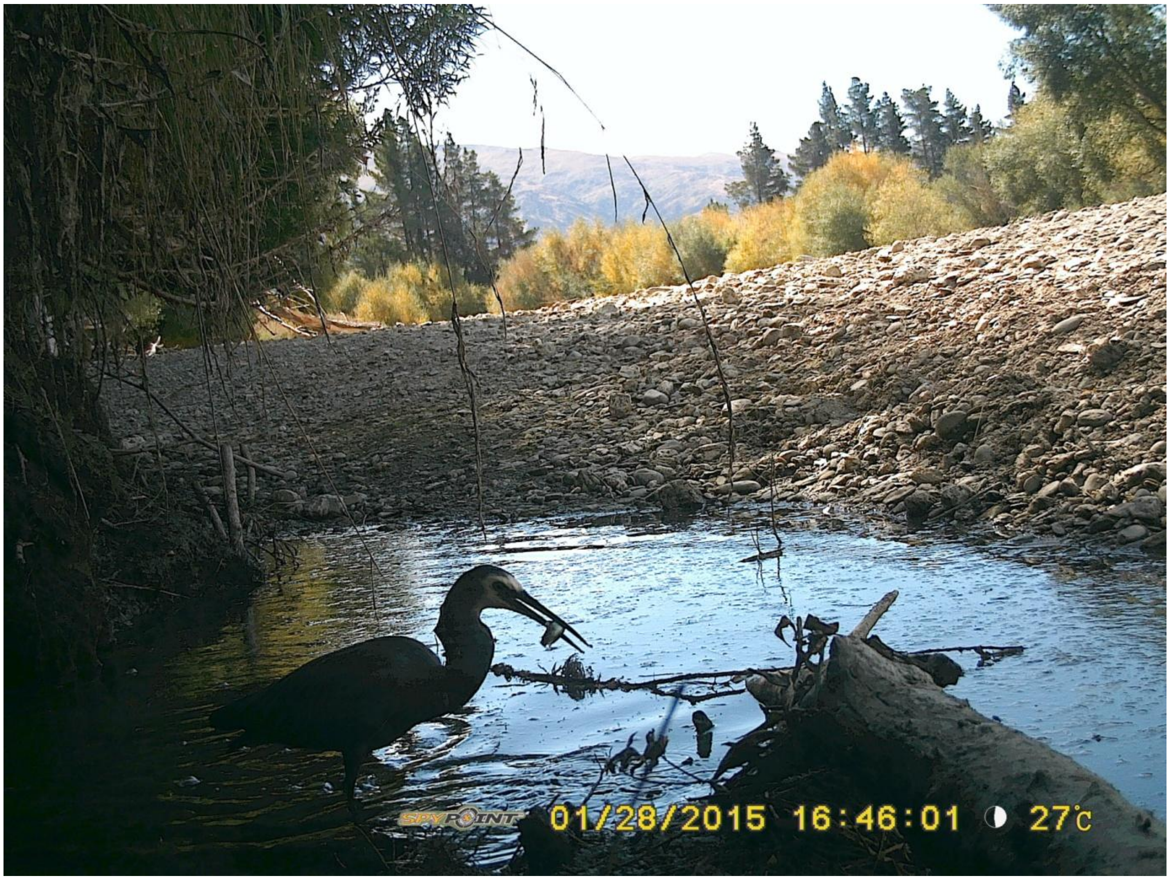

Figure 2.6. A white faced heron feeding on one of the few remaining trout stranded within a drying pool 28 January 2015. Initially there were 30 PIT sample fish alive in this pool. Two days after this photo was taken the pool dried entirely and one loose PIT tag was found.

2.4.5.2 Water Temperature and Dissolved Oxygen

The temperature logger deployed in the higher stress reach was completely de-watered within 13 days and was operating in shallow standing water prior to dewatering. The observed water temperatures varied between 12.87 °C and 24.22 °C and averaged 17.69 °C. The highest temperatures observed occurred before dewatering of the logger, which may have been affected by solar radiation (Johnson and Wilby 2013). The logger deployed within the intermediate stress segment remained in flowing water and functional throughout the study period. Temperatures recorded within this segment varied between 11.27 °C and 17.17 °C and averaged 13.59 °C.

The lowest recorded dissolved oxygen readings (24.10 % and 2.67 mg/l) were taken in flowing water located in the intermediate stress segment in the bed of a deep pool in relatively cool water (9.90 °C). The dissolved oxygen and temperature readings at the surface of that pool at that time were higher (12.8 °C, 60.6 % and 6.28 mg/l). The highest temperature (21.00 °C) was recorded within the higher stress segment.

Table 2.5 Dissolved oxygen (DO % saturation and mg/l) and water temperature measurements from pools with flowing water in the higher and intermediate stress segments in year two.

Site	Pre-Dawn DO %			Afternoon DO %		
	Mean	Max	Min	Mean	Max	Min
Intermediate stress	53.06	60.70	24.10	80.72	90.30	62.10
Upper higher stress	84.24	84.90	81.70	108.48	108.70	108.30
Lower higher stress	84.71	86.50	79.60	111.23	113.40	103.20

Site	Pre-Dawn DO mg/l			Afternoon DO mg/l		
	Mean	Max	Min	Mean	Max	Min
Intermediate stress	5.55	6.29	2.67	7.96	8.68	6.68
Upper higher stress	8.62	8.68	8.36	10.32	10.36	10.28
Lower higher stress	8.41	8.61	7.89	9.80	9.91	9.38

Site	Pre-Dawn temp °C			Afternoon temp °C		
	Mean	Max	Min	Mean	Max	Min
Intermediate stress	12.21	12.80	9.90	14.93	16.30	11.20
Upper higher stress	13.40	13.40	13.40	16.77	16.90	16.60
Lower higher stress	14.84	14.90	14.70	20.58	21.00	19.00

2.4.6 Physical habitat analysis

Results of the two-sample Kolmogorov-Smirnov test found the percentage area of variables that reflect juvenile trout cover (woody debris, submerged branches, overhanging vegetation, undercut bank) was significantly greater in the intermediate stress segment than in the higher stress segment (Table 2.6). The intermediate segment also contained significantly a greater area of water 30 - 50 cm deep, while the higher stress segment contained significantly larger area of water <30 cm deep (Table 2.6). The reach area, mesohabitat types and most substrate comparisons were not significantly different between the higher and intermediate stress segments (Table 2.7), although there was significantly more small cobble habitat in the higher stress segment than intermediate segment (Table 2.6).

Table 2.6 Results of a two-sample Kolmogorov-Smirnov test displaying significant differences in physical habitat variables between higher and intermediate stress stream segments.

Variable	Mean		Statistic	*P*-Value
	Higher stress	Intermediate stress		
Percent of Depth < 30 cm	91.111	64.722	0.556	0.008
Percent of Depth 30-50 cm	5.556	25.500	0.611	0.002
Percent of Small cobble	24.583	15.278	0.444	0.057
Percent area of woody debris	0.154	0.516	0.444	0.057
Percent area of submerged branches	0.066	1.584	0.667	0.001
Percent area of overhanging vegetation	0.025	0.429	0.444	0.057
Percent area of undercut bank	0.118	0.743	0.611	0.002
Total area of all cover types	0.220	2.687	0.722	0.000
Percent area of all cover types	0.082	1.226	0.722	0.000

Table 2.7 Results of a two-sample Kolmogorov-Smirnov test displaying non-significant difference in physical habitat variables between higher and intermediate stress stream segments.

Variable	Mean		Statistic	*P*-Value
	Higher stress	Intermediate stress		
Reach area	239.815	239.556	0.333	0.270
Percent Riffle	46.944	33.333	0.222	0.766
Percent Slow run	27.778	30.833	0.111	1.000
Percent Fast run	0.000	0.000	0.000	1.000
Percent Pool	25.833	35.833	0.167	0.964
Percent of Depth 50-100 cm	2.222	5.833	0.222	0.766
Percent of Depth >100 cm	1.111	3.889	0.111	1.000
Percent of Fine sediment	10.833	13.333	0.389	0.131
Percent of Gravel	13.056	29.722	0.389	0.131
Percent of Coarse Gravel	22.639	23.056	0.111	1.000
Percent of Large cobble	24.167	12.222	0.333	0.270
Percent of Boulder	0.556	5.833	0.167	0.964
Percent of Bedrock	0.000	0.000	0.000	1.000
Percent of Macrophyte cover	5.500	0.000	0.167	0.964
Percent area of turbulence	0.000	0.587	0.222	0.766
Residual pool depth m	0.652	1.310	0.667	0.108

The correlation-based PCA revealed two main axes accounting for 64.9 % of the total variance in habitat variables between stream segments (Figure 2.7). The first factor was highly correlated with total area of cover, and percent area of cover, and percent area of submerged branches (Table 2.8). The second factor reflected water depth (percent area 0 - 0.3 m and percent area 0.3 - 0.5 m) and percent area of undercut banks. There was a significant difference in the habitat within higher and intermediate stress stream segments (ANOSIM R=0.239, P=0.001).

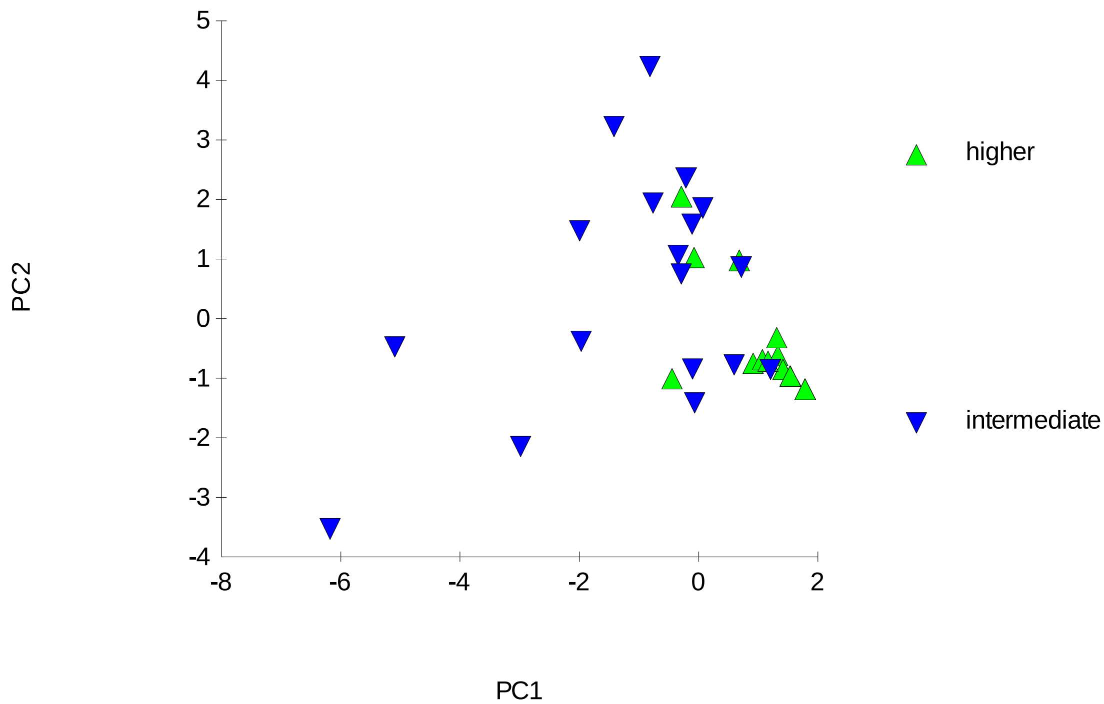

Figure 2.7 Plot of the factor scores for habitat variables on the first two principal components for higher and intermediate stress stream segments. PC 1 (total area of cover, percent area of cover, percent area of submerged branches) and PC 2 (percent area 0-30 cm depth, percent area 30-50 cm depth) account for 64.9% of the total sample variability.

Table 2.8 Factor loadings for the first two principal components from PCA analysis of habitat variables in higher and intermediate stress stream segments.

Variable	PC1	PC2
Depth 0-0.3 m	0.179	**-0.558**
Depth 0.3-0.5 m	-0.183	**0.471**
Small cobble	0.318	-0.281
Percent area of undercut banks	-0.114	**0.445**
Percent area of overhanging vegetation	-0.287	0.056
Percent area of woody debris	-0.301	0.017
Percent area of submerged branches	**-0.432**	-0.233
Total area of cover	**-0.483**	-0.253
Percent area of cover	**-0.478**	-0.26
Variation explained (%)	37.0	27.9

2.5 Discussion

Mark-recapture analysis indicates that the juvenile salmonid population of the lower Lindis River experience extremely low survival during the low flow abstraction period. In year one flows reduced to approximately 25 % of MALF or less for 70 days and the survival rate was estimated at 0.34 (95 % CI: 0.21-0.50). In year two flows remained at approximately 25 % of MALF or less for the entire period of 59 days, and survival was estimated at 0.29 (95 % CI: 0.18-0.39). These survival rates are much lower than would be expected for post-critical period juvenile salmonids (Elliott 1994, Jonsson and Jonsson 2011) and lower than found in the Rainy River population where mortality was estimated at 29 % over a nine month period (Holmes et al. 2013). Mortality rates of 80 - 90 % can be expected during the early emergence critical period (Elliott 1994, Jonsson and Jonsson 2011, Quinn 2011). Mark-recapture analysis shows that post-critical period, juvenile salmonids in the Lindis River are subject to an additional period of high mortality (approximately 70 %) during the summer low flow period. The effects of extreme low flow events can result in long term impacts on salmonid population structure (Elliott 1993, Elliott et al. 1997, Jonsson and Jonsson 2011) and it is likely that recruitment potential of Lindis juveniles to other waters is limited by these low flow events (Gabrielsson 2015).

Stationary cameras revealed that when Lindis River fish are subjected to low flow events they experience high levels of avian predation. Reduced flows in the higher stress segment were associated with the lowest levels of cover for juvenile trout, such as submerged and over hanging vegetation and woody debris (Vehanen et al. 2000). A loss of this type of cover can increase stream fish predation (Crook and Robertson 1999, Magoulick and Kobza 2003) and reduced cover was associated with the lowest survival rates. The higher stress segment also contained a higher percentage of habitat less than 30 cm deep, and it is likely these conditions would favor predation on trout by herons (Hodgens et al. 2004). The duration of the low flow period during both years of my study (70 days in year one, and 59 in year two) meant that juvenile trout were subjected to high levels of predation for extended periods. This would explain the low survival of fish in the study reach.

Water temperatures taken from flowing water within the Lindis River remained within trout tolerance levels and it is therefore unlikely that fish generally died from thermal stress. Salmonid survival during droughts is dependent on suitable temperature and oxygen concentrations being available in pools, with fish demonstrating localised movement to avoid exceeding thermal limits (Matthews and Berg 1997, Elliott 2000). The incipient lethal temperature (where 50% of the population may survive for 7 days but not indefinitely) for brown and rainbow trout has been estimated at approximately 25°C (Elliott 1981, Matthews 1997). The ultimate lethal temperature for brown and rainbow trout has been estimated at approximately 28 - 29°C (Elliott 1981, Matthews and Berg 1997). The response of trout to high temperatures depends on the temperatures they have been acclimatised to (Elliott 1994). Fish kills from heat stress in Otago streams primarily occur when they become stranded in shallow pools isolated from the main flow (Caruso 2001).

Monitoring of marked fish stranded in pools revealed that active predation reduced their numbers, while un-predated fish survived the physical conditions found within the pools. This is further indication that predation and not heat stress was the primary cause of their demise. Stream dissolved oxygen concentrations vary depending on water temperatures and ground water seeps, and levels of 1.7 to 3.4 mg/l are considered to be potentially lethal to salmonids (Matthews and Berg 1997). One reading of low dissolved oxygen at the bed of a pool in the Lindis River (24.1 %, 2.67 mg/l) was attributed to ground water seepage as the water was cooler at the bottom (9.9 °C) than surface (12.8 °C). Oxygen levels at the surface of that pool were within trout tolerance levels (60.6 %, 6.28 mg/l). Despite regular and thorough monitoring, none of the 1016 fish marked during the course of this study were found dead. All of the 16 fish that were marked after death were removed by scavengers overnight. These results indicate that any dead fish observed during prolonged low flow conditions are likely to represent a small proportion of the total mortality experienced.

2.5.1 Model assumptions

It is important to ensure that any major assumptions of a survival model are reasonable (Lettink and Armstrong 2003). The CJS model assumed that all fish not encountered by the end of the study period were dead. A large proportion of PIT tags employed in this study were not redetected (464/1016) and detection rates declined over the course of both years; accounting for these missing tags is an important consideration. There are four likely scenarios.

i. Marked fish died due to tagging related mortality, or shed their tags and remained alive

The use of PIT tags to study juvenile salmonids is well documented and studies have shown that rates of mortality and tag shedding by live fish are low, provided the fish are of an appropriate size (Acolas et al. 2007, Richard et al. 2013b). Precautionary size limits and a single PIT tagger who had demonstrated correct PIT tagging technique at the Otago University were employed to mark all sample fish. Furthermore, significant mortality related to PIT tagging was unlikely because the recovery of all captured and tagged fish was confirmed before release (Berger and Gresswell 2009).

ii. Sample fish moved out the study area alive

Fish movement along the river was constrained by the dry river bed downstream and upstream of the study area during both years. In year one the riverbed downstream of the study area dried during the start of the initial tagging period. There was only a brief window when some fish movement to the Clutha River may have been possible. Tagging began upstream of the intermediate fixed antenna, which indicated 5 sample fish detections during this time frame. Three of these fish were later redetected indicating that they did not leave the study area. The other two fish were not redetected so their fate remains unknown. While there is a chance that these fish may have migrated out of the study area alive, this is not considered to be a significant source of bias as the relative proportion of fish in question is low (2/1016). The reconnection of surface flow at the end of year one occurred after the last major mobile survey and would not have affected results as low flow related impacts would have already occurred. The irrigation channel was not a conduit that could allow sample fish to leave the study area alive. Any fish that become entrained in Lindis irrigation channels end up in paddocks or become stranded after the scheme is turned off for winter each year (Jellyman and Bonnett 1992) and die. In year one only 7 sample fish were recorded in the irrigation channel. In year two, sample fish could not access the irrigation channel. Thus, there is a high level of confidence that the survival estimate represents actual mortality, as opposed to observed mortality (or apparent survival), whereby a significant proportion of the sample may have migrated from the study area alive (Berger and Gresswell 2009).

iii. PIT tags remained within the study area but were not located

McEwan and Joy (2011) noted that the physical complexity of an open river meant that a PIT tagged fish or a tag may be missed. If a large proportion of tagged fish or shed tags were not encountered by monitoring (McEwan and Joy 2011) a mortality estimate would be overly high. This was accounted for by undertaking 13 field tests of mobile reader detection efficiency to obtain a detection efficiency value which was used to adjust CJS model survival estimates.

iv. PIT tags were removed from the study area

Of the 51 PIT sample fish that were stranded in residual pools subject to predation, only one loose tag was located after the pools dried. Of the 16 dead PIT tagged trout placed in a residual pool and subjected to scavenging overnight, no marked fish or PIT tags could be found by the following morning. Out of 1016 fish initially tagged only 29 loose tags were found on the riverbed, and the deaths of the fish into which 20 of these tags had been placed could be attributed to black shag predation. These results indicate that a very high proportion of PIT tags were removed from the study area by predators and scavengers.

v. Individuals have an equal chance of being detected

Another important assumption of the model is that all individuals had an equal chance of being detected. The survival model analysis in year one indicated that there was no difference in estimated survival rates between species (brown or rainbow trout) and therefore detection probability was likely to be similar. Ten percent of sample fish were 1+, tagged with larger tags which had an improved read range. Seven percent of redetections events were from 1+ fish. While the behaviour of 1+ trout may increase their chance of evading mobile antenna detection (Cucherousset et al. 2005), the increased size of PIT tags employed in 1+ trout as opposed to 0+ may be expected to reduce the difference in probability of detection to some extent. Whilst this remains a limitation of my study, it is not considered to be a major source of bias as the initial component of 1+ fish tagged was relatively low (105/1016 fish tagged). Also the detection efficiency trials used to develop a detection efficiency estimate and adjust final survival estimates was the same ratio of large to small tags (ten percent) as employed in sample fish.

The limited detection range of the fixed antenna meant that that mainstem antenna performance was compromised at flows greater than approximately 2.5 m^3/s, and fish moving at flows above this level would experience a lower chance of being detected. Flows exceeded 2.5 m^3/s at the end of the year one after the last major mobile survey. The movement of remaining sample fish at this time would not have affected the survival estimate, as low flow related impacts on survival would have already occurred.

2.5.2 Management implications and future research

Abstraction of water for irrigation can compound the effect of natural low flow events on aquatic ecosystems. Despite the longevity of the problem the impacts of over-abstraction are commonly seen as fragmented river habitat and degraded fish populations (Bond et al. 2008). Most previous estimates of juvenile trout mortality related to low flow events have been speculative as they were unable to assign fish losses to movement or mortality (Holmes et al. 2013). Mark recapture analysis and subsequent physical habitat and predation monitoring in the Lindis River has shown that current levels of abstraction and low flow conditions are resulting in high juvenile trout mortality rates. This is primarily due to lack of habitat cover from predation. Streams that dry to isolated pools and dewatered riverbed are likely to act as populations sink areas, where annual recruitment is lost (Magoulick and Kobza 2003). It is expected that the high mortality of Lindis River juvenile trout would reduce the recruitment potential of this population to other waters such as the Upper Clutha River. These findings should be taken into account when considering minimum flow options for the Lindis River. It is expected that similar patterns of high predation related mortality occur in other heavily abstracted streams. It is also probable that habitat models that do not consider the relationship between reduced flows and limited predation cover for juvenile trout are overly simplistic. These points suggest promising avenues for future research to improve flow management and protection of stream ecosystems.

Chapter 3: Movement of juvenile trout under summer low flow conditions in the Lindis River

3.1 Abstract

Movement of juvenile brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) was investigated during low flow conditions in the Lindis River of Central Otago, New Zealand. Significant abstraction from this river results in drought-like events on an annual basis. Over two years, during the summer low flow period >1000 juvenile trout (age 0+ and 1+) were marked with Passive Integrated Transponder (PIT) tags. After PIT tagging, the study area was subjected to extreme low flow conditions in both years of <25% of mean annual low flow (MALF) for 75 days in year one, and <25 % of MALF or less for the entire 59 day study period in year two. To detect migration to potential refuge habitat, such as the Clutha River, stationary antenna were installed in the mainstem of the Lindis River. To detect if fish moved from stressful drying habitat to potential refuge habitat, such as remaining pools or river segments where surface water connection was maintained, their movements were tracked using mobile PIT tag antenna. To detect if fish were lost to an irrigation channel, stationary PIT tag antenna were positioned in the channel. Potential fish movement to refuge habitat was constrained by areas of dry riverbed, and an irrigation intake structure. Fish that were released into stream segments that experienced the least loss of surface water and drying were more likely to stay in that habitat, than those released into segments that experienced more stressful conditions ($P < 0.01$). Whilst there was no indication of a large scale migration attempt during low flow conditions, some sample fish displayed considerable movement along the remaining

fragmented river corridor. This was presumably an attempt to leave stressful habitat. During the extreme low flow events juvenile trout preferred shallow riffle habitat to deeper pools.

3.2 Introduction

3.2.1 Triggers of juvenile trout migration

Instinctual behaviour and environmental factors are thought to influence the timing of juvenile trout migration from their natal stream (Elliott 1994, Jonsson and Jonsson 2011). The most important environmental factors appear to be temperature and flow when the fish are ready to move (Jonsson and Jonsson 2011). It is often difficult to differentiate what factors, or combination of factors trigger migration (Kahler and Quinn 1998).

Density dependent competition for food and habitat has been suggested as an important trigger for migration in some juvenile brown trout populations (Armstrong et al. 2003, Milner et al. 2003, Jonsson and Jonsson 2011). Fast growing juvenile trout (Acolas et al. 2012) and those approaching growth restraints (Jonsson and Jonsson 2011) are often the first to migrate. This is presumably because they are seeking better feeding grounds in order to maintain their high metabolic rate (Forseth et al. 1999). Migration of fast growing fish suggests that competition for food and habitat via a density dependent process may encourage self-thinning of the population (Lobón-Cerviá and Mortensen 2006). It is still not understood why some juveniles leave their natal rearing stream and others stay (Elliott 1994). Studies in a small New Zealand stream found that migrating fry were a similar size to some that stayed, indicating that density dependent competition for habitat could not solely explain migration (Kristensen 2006).

Migration events can also be triggered by adverse habitat conditions such as increasing water temperatures and declining flows (Kahler et al. 2001, Jonsson and Jonsson 2011). In some instances a reduction of habitat experienced during drought and low flows may increase competition and cause inferior competitors, such as the smaller fry, to migrate (Landergren 2004). Some juvenile sea-run brown trout populations appear to have evolved a migration strategy of leaving their natal rearing stream at an unusually early time and small size (< 8 cm) in advance of drought and low flow impacts (Borgstrøm and Heggenes 1988, Jonsson et al. 2001, Landergren 2004).

3.2.2 Juvenile brown trout migration patterns

Juvenile trout migration from their natal stream to adult habitat typically involves large scale one-way behavioural movements, often downstream under the cover of darkness (Elliott 1994, Jonsson and Jonsson 2011). After emergence from gravels, most brown trout alevins (approximately 80 %) are in poor condition and drift downstream at night and die (Elliott 1994). Survivors establish feeding territories near their nest, some may migrate downstream a short distance before they find refuge, often under a large cobble (Elliott 1994). Density dependent competition for feeding territory may encourage this movement (Elliott 1994, Jonsson and Jonsson 2011). Some brown trout fry may choose to remain in their natal rearing stream to become resident fish, while others migrate to larger waters presumably to increase growth and fecundity potential (Kristensen and Closs 2008, Jonsson and Jonsson 2011).

Brown trout display a remarkable range of life history adaptations (Jonsson and Jonsson 2011). The plasticity of the species may best explain differences in juvenile migration patterns that have been reported. In New Zealand migration of juvenile trout has been recorded in all months of the year. Migrations of young of the year (0+) and yearling (1+) fish in the Rakaia River generally peak in spring (see review in Jellyman and Bonnett 1992). In the Rainy River most migration of juveniles occurs during autumn and spring freshes, and to a lesser extent during spring low flows (Holmes et al. 2013). In the lower Silver Stream juvenile brown trout cohorts that were not displaced by high flows, left their natal rearing stream during summer low flow conditions (Kristensen and Closs 2008).

3.2.3 Juvenile rainbow trout migration patterns

After emergence most rainbow trout alevins are thought to stay near their redd and try to establish territories or disperse downstream (Quinn 2011). Natal stream residence is a common life history adaptation in juvenile rainbow trout (Rosenau 1991) , but in some streams rainbow trout are more likely to migrate after emergence than brown trout (Hayes 1988a). While early migrating fry experience much higher mortality than late migrants, a proportion of early migrating rainbow fry can survive in a lake system provided there is favorable juvenile rearing habitat such as overhanging vegetation, submerged branches and weed beds (Hayes 1995a). In Lake Taupo any small recently emerged fry that enter the lake experience extremely high levels of mortality (Rosenau 1991). Juveniles need to remain in their rearing stream until they are at least 94 mm to have much chance of survival (Rosenau 1991) . Low survival of newly emerged alevins in larger waters may be due to their inability to find food of a suitable size (Rosenau

1991) , or compete for food resources with other small fishes (Rosenau 1991, Hayes 1995a). In New Zealand out-migration of 0+ rainbow trout from nursery streams has been recorded to commence in October and peak during November to January (Hayes 1988b, Graybill and Palmer 1990). Additional smaller migrations of juveniles have also been recorded in autumn and early winter when spawning adults entered the stream (Hayes 1988b). Major floods have been found to increase migration but smaller freshes seem to have little effect (Graybill and Palmer 1990).

3.2.4 Juvenile trout movement and migration during low flows

Juvenile trout are known to exhibit both small scale movements and large scale migration to avoid stressful conditions during drought and low flows (Elliott 2000, see review in Hay 2004, Landergren 2004, Jonsson and Jonsson 2011). There is substantially more understood about smaller scale movements under low flow conditions. Juvenile salmonids will often move from shallow riffles and runs to deeper pools, seeking cool water refuge (Campbell and Scott 1984, Huntingford et al. 1999, see review in Hay 2004). While pools are generally considered to be the preferred refuge of juvenile trout from high water temperatures experienced during drought conditions (Elliott 2000), some investigations have found that riffle habitat can be preferred (Davey and Kelly 2007, Ayllón et al. 2009). This is possibly because riffle habitat can provide cover from predation (Henderson and Letcher 2003). Juvenile salmonids most commonly move upstream to seek refuge in response to declining flows and increasing temperature (Gowan and Fausch 1996, Kahler et al. 2001, Davey and Kelly 2007). For example, in one experimental stream where flows were reduced, 20 % of juvenile salmonids did not move, and of those that did move between 74 - 89 % went upstream, with the remainder moving downstream (Huntingford et al. 1999).

Greater understanding of population response to low flows is needed, particularly where natural flow regimes are altered as a result of abstraction and managers are faced with determining appropriate minimum flow to support instream values (Hayes et al. 2010, Holmes et al. 2013). Small scale movement to refuge habitat such as pools has been suggested to mitigate juvenile salmonid losses in streams subjected to high levels of abstraction (ORC 2008). Juvenile salmonids may also undertake larger scale movements from stressful habitat, within their natal stream or to a larger water body (Gowan and Fausch 1996, Kahler et al. 2001, see review in Hay 2004). Complete drying of the riverbed may prevent fish passage to refuge habitats and result in fish strandings (Davey and Kelly 2007). Impeded fish passage can also affect natural migration patterns (Elliott 2000), so it is also important to understand how altered flow regimes might impact juvenile trout life history.

To investigate juvenile trout movement in the Lindis River which is subject to significant abstraction (Jellyman and Bonnett 1992, ORC 2008), >1000 young of the year (0+) and yearling (1+) trout marked with Passive Integrated Transponder (PIT) tags were tracked over two consecutive summer low flow periods. The objectives were to (i) determine if juvenile trout moved from stressful habitat to refuge habitat during low flow events and, (ii) determine if juvenile trout migrated from the Lindis River to the Clutha River during the summer low flow period.

3.3 Methods

3.3.1 Study Area

The Lindis River is a tributary of the Upper Clutha River and is located in the arid Central Otago region of New Zealand. The lower Lindis valley is one of the driest regions in New Zealand with an average annual rainfall of approximately 500 mm, and as little as 300 mm some years. The river experiences very high levels of abstraction for irrigation which results in the complete drying of approximately 10 km of lower river reaches in most summers. The surface flow reduction is compounded by groundwater losses in affected reaches (Jellyman and Bonnett 1992, ORC 2008). The estimated naturalized (without abstraction) mean annual low flow (MALF) of the lower Lindis is 1.860 m^3/s. However, after abstraction for irrigation the actual MALF is 0.177 m^3/s (Horrell 2014). The Lindis River is considered to be an important spawning and nursery stream contributing recruitment to the Upper Clutha River and Lake Dunstan trout fisheries (Jellyman and Bonnett 1992, Turner 1994). The Lindis predominately contains 0+ and 1+ juvenile brown and rainbow trout, with brown trout being the most common species (Jellyman and Bonnett 1992). For more detailed information on the study area see section 1.4.

3.3.2 Fish capture, passive integrated transponder tagging and tracking

To investigate patterns of juvenile trout movement during the low flow period the PIT tag mark-recapture dataset was re-examined. A brief summary of the field methodology used to obtain the dataset is provided here. For a detailed explanation of fish capture, tagging and tracking methodology see section 2.3.

The study area was divided into three stream segments (Figure 2.1) based on observed differences in surface water connection, which are due to variation in ground water losses along the lower river (Jellyman and Bonnett 1992, ORC 2008). The higher stress segment experiences significant losses of surface water to groundwater (Jellyman and Bonnett 1992, ORC 2008) and approximately 50 % of this segment dries up completely during the summer abstraction period. Approximately 10 % of the intermediate stress segment riverbed dries up as a result of an irrigation channel diversion arm which diverts the surface water and dewaters the riverbed immediately downstream. In the lower stress segment surface flows are significantly reduced but surface water connection is maintained. The irrigation channel (Figure 2.1) is a priority take and upstream abstractors must deliver a consistent amount of water to this point (Jellyman and Bonnett 1992).

In each of the stream segments juvenile trout were captured, PIT tagged, and released. Movement of sample fish was then monitored using fixed and mobile antennas over two summer-autumn low flow periods (year one and year two). The fixed antennas were deployed in the mainstem river (Figure 2.1) to continuously monitor for downstream movements which might be associated with outmigration. An additional stationary antenna was located in an irrigation channel to account for any losses to the channel (Figure 2.1). To determine movement patterns across a large spatial scale the year one investigation included approximately 9 km of riverbed, comprising the higher, intermediate and lower stress segments. In year two the spatial scale was reduced to the higher and intermediate segments only (approximately 3 km of riverbed). This allowed more intensive monitoring, and six complete mobile surveys of the study area were completed in year two, as opposed to three in year one. Flow data was obtained from the Otago Regional Council flow recorder located in the intermediate stress segment (Figure 2.1). To gain information on preferred habitat during low flow conditions the mesohabitat type (pool, run or riffle) and depth of each fish detection was also recorded in year two.

3.3.3 Data analysis

3.3.3.1 Movement from stressful habitat

To investigate if juvenile trout moved from stressful habitat the fish movement pattern between the three stream segments which experienced varying levels of flow reduction and drying were investigated. The fish movement pattern was analysed by examining the unique number of individuals found during each survey period (based on mobile and fixed antenna survey results) in each stream segment, relative to the total number of fish initially released at each stream segment. A Chi-square test was applied to test the hypothesis that the overall proportions of fish detected are equal among habitats for fish released at each habitat. P-values from the pairwise chi-square tests were adjusted with Bonferroni correction.

3.3.3.2 Migration

To investigate if sample fish migrated from the Lindis River to the Clutha River the dataset was examined for a relatively large number of movements at mainstem antenna over a short period. This would be indicative of a juvenile trout migration event (Holmes et al. 2013). Juvenile trout migrating from their natal rearing stream often move during freshes (Holmes et al. 2013) so the relationship between mainstem antenna detections and river flow patterns was also examined by conducting a regression analysis.

3.3.3.3 Irrigation Channel Losses

To determine if sample fish were lost to an irrigation channel that was open to the river the dataset from the fixed antenna installed within the irrigation channel in year one was examined.

3.3.3.4 Fish size and movement

To determine if there was a difference in the distance moved between 0+ and 1+ during low flow conditions, a t-test was conducted for the null hypothesis that there is no significant difference in mean distance travelled between 0+ and 1+ fish. This was measured using individual fish detection locations to determine the distance they had moved during each survey period. This analysis was conducted for each survey period to reduce temporal variation between individual fish detections. Analysis included data from year one only because the sample size of 1+ fish in year two was considered insufficient (21/394 fish). Fish ≥130 mm were considered likely to be 1+ fish, based on previous studies of the trout population in the Lindis River (Jellyman 1990).

3.3.3.5 Mobile fish

To further explore patterns of movement in response to low flow conditions the detection histories of the 20 most mobile fish in both years were plotted in order to show the direction (upstream or downstream) and distance moved from their release point.

3.3.3.6 Mesohabitat Preference

To determine if juvenile trout moved to refuge pools during summer low flow conditions (Elliott 2000) the mesohabitat location (pool, run or riffle) and depth of each fish detected during mobile antenna surveys (n = 460) were recorded in year two.

3.4 Results

3.4.1 Capture and tagging

The capture and tagging period extended from 21 January to 5 February 2014 in year one, and from 7 to 13 January 2015 in year two. The details of the fish sampled, and the sample locations are summarised in Table 2.4.

3.4.2 Passive integrated transponder tag encounters

Over both years, there were a total of 1485 detection events from 565 tags. The majority (1052) were recorded during mobile antenna surveys. All of these were confirmed to be from live fish except for 29 loose PIT tags that were found on the bed of the river and assumed to be from fish that had died. In year one the irrigation channel antenna recorded eight detections from seven tags, and the antenna located within the intermediate stress segment recorded 31 detections from 15 tags. In year two the antenna located within the higher stress segment recorded 67 detection events from 39 tags. The antenna located within the intermediate stress segment recorded 327 detection events from 100 tags.

3.4.3 Movement restrictions

Possible movement to the Clutha River and the upper Lindis River was constrained by areas of dry riverbed in both years of the study (Figure 3.1). In year one the lower river dried soon after the start of the tagging and release period, and there was only a brief window when fish may have potentially been able to access the Clutha River. The initial tagging and release of sample fish began upstream of the intermediate antenna so that any downstream movement of sample fish would be detected. Downstream movement of 5 sample fish was detected during the period when fish could have accessed the Clutha River (Figure 3.2). Three of these fish were subsequently redetected indicating they had not left the study area; the other two were not detected again. An area of dry riverbed 3 km upstream of the study area prevented upstream movement of fish from the study area. To confirm that sample fish had not moved upstream out of the study area a mobile PIT tag reader survey started at the dry riverbed upstream of the study area. No sample fish were found more than 200 m from the study area. Two subsequent mobile surveys started 700m upstream of the study area, and also found no sample fish more than 200m from the study area.

In year two the riverbed downstream of the study area was dry for the entire study period and an irrigation diversion arm, which diverts almost all surface flow into an irrigation channel during low flow conditions prevented upstream passage of fish out of the study area. This was confirmed by mobile antenna surveys, starting up to 2.3km upstream of the study area.

As the summer low flow period progressed, drying of the riverbed within the intermediate and higher stress segment also restricted potential fish movement. Approximately 10 % of the intermediate stress segment downstream of irrigation diversion arm dried in both years. Approximately 50 % of the higher stress segment dried entirely in both years (Figure 3.1). In year two 51 live sample fish were constrained in drying pools located within the higher stress segment.

Figure 3.1 Dewatering of the higher stress segment 15 March 2014. Drying riverbed can be seen in the middle of the picture, as well as two disconnected pools in the background.

3.4.4 Fish movement between stream segments

Year One

For fish released in the lower stress segment there were significant differences in the proportion of fish later detected in other stream segments (*P*<0.001) (Table 3.1). Further analysis (pairwise chi-square test) showed that these differences were all due to the proportion of fish staying within the lower stress segment. This indicates that fish released at the lower stress segment were most likely to stay within that segment.

For fish released in the intermediate stress segment, there were no significant differences ($P > 0.05$) found in the proportion of fish detected in the three stream segments for all of the survey intervals, suggesting significant movement from the intermediate stress segment across other stream segments.

For fish released into the higher stress segment, significant differences ($P < 0.01$) were found in the proportion of fish detected in the three stream segments. Further pairwise chi-square analysis for the third occasion shows that the difference between higher stress and intermediate stress is not significant ($P > 0.05$), indicating some significant movement from higher stress segment to intermediate segment. There was no fish movement from the higher stress segment to the lower stress segment.

Table 3.1 Summary of fish released, and detected in each stream segment for each survey period in year one.

Release habitat	Release number	Survey period	Detection Habitat		
			Lower stress	Intermediate stress	Higher stress
Lower stress	488	1	99 (20%)	0 (0%)	7 (1%)
		2	127	1	0
		3	102	7	3
Intermediate stress	48	1	2 (4%)	4 (8%)	2 (4%)
		2	2	2	1
		3	2	2	0
Higher stress	86	1	0	1 (1%)	27 (31%)
		2	0	3	18
		3	0	3	7

Year Two

Of the fish released into the intermediate stress segment, a small proportion was later detected in the higher stress segment (Table 3.2). A two-sample test for equality of proportions (with continuity correction) shows that there were significant differences in the proportion of fish detected in the two habitat types ($P < 0.01$) for all detection occasions. This test result suggests that although fish could move from intermediate stress segment to the higher stress segment, they were more likely to stay in the released intermediate stress segment.

Of the fish released into the higher stress segment, the relative proportion of fish detected at intermediate stress segment (compared to the higher stress segment) increased with each detection occasion (Table 3.2). The two-sample test for equality of proportions shows that significant differences in the proportions were found until detection occasion 3 ($P < 0.01$). On occasion 4, however, the difference was not significant ($P > 0.05$). These results suggest a significant movement from higher stress segment to intermediate stress segment in the first detection occasions.

Table 3.2 Summary of fish released and detected in each survey period at each stream segment in year two.

Release habitat	Number of fish tagged	Survey period	Detection Habitat	
			Intermediate stress	Higher stress
Intermediate stress	91	1	85 (93%)	6 (7%)
		2	58	6
		3	29	2
		4	19	0
Higher stress	303	1	22 (7%)	281 (93%)
		2	19	171
		3	10	29
		4	7	8

3.4.5 Fish movement and river flow relationships

Year One

During year one river levels dropped to approximately 25% of MALF soon after tagging and remained at this level for 75 days. The total number of fish that moved past the intermediate segment antenna in year one was relatively low (n = 28). Regression analysis found there was no indication of a relationship between stream flow and detections at the intermediate segment antenna (R^2= 0.023) (Figure 3.2). In total only eight detections were recorded at the irrigation channel from seven individual fish (Figure 3.3). Four of these detections occurred during freshes after the low flow period had ended, and a positive relationship was found between increasing flow and fish movements ($R^2 = 0.570$) (Figure 303).

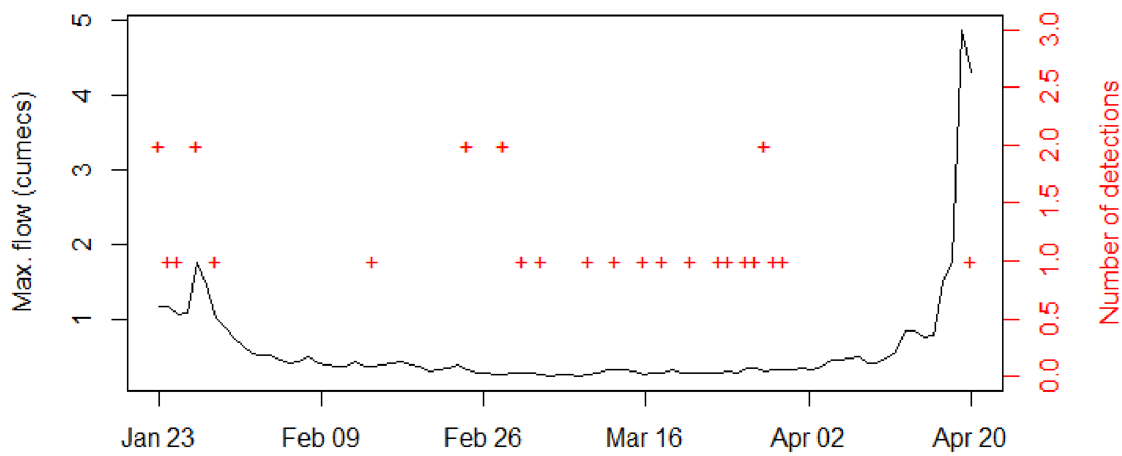

Figure 3.2 *Maximum daily flow (m3) and intermediate stress segment antenna detections during year one*

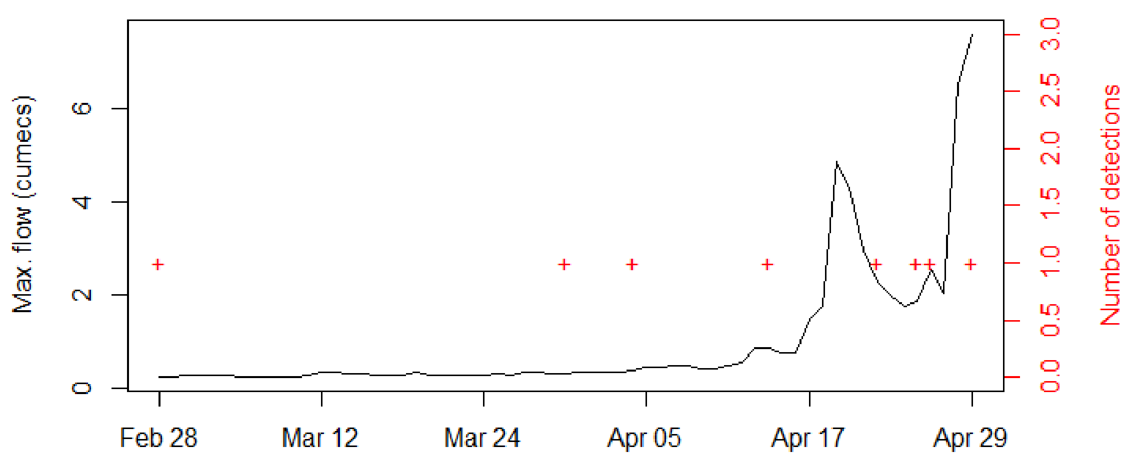

Figure 3.3 Maximum daily flow (m3) and irrigation channel antenna detections during year one

Year Two

In year two river levels remained at approximately 25 % of MALF or less for the entire study period. There was a weak relationship between decreasing flow and fish movements at the higher stress antenna (R^2=0.31) (Figure 3.4). There was not enough water at the stationary antenna after 16 January for fish to be physically able to move past it. It became totally dewatered on 19 January. There was no relationship between stream flow and detections and the intermediate stress segment in year two (R^2=0.005) (Figure 3.5). The most movement past the intermediate stress segment antenna occurred January 12-13 when 72 individual fish moved past the antenna.

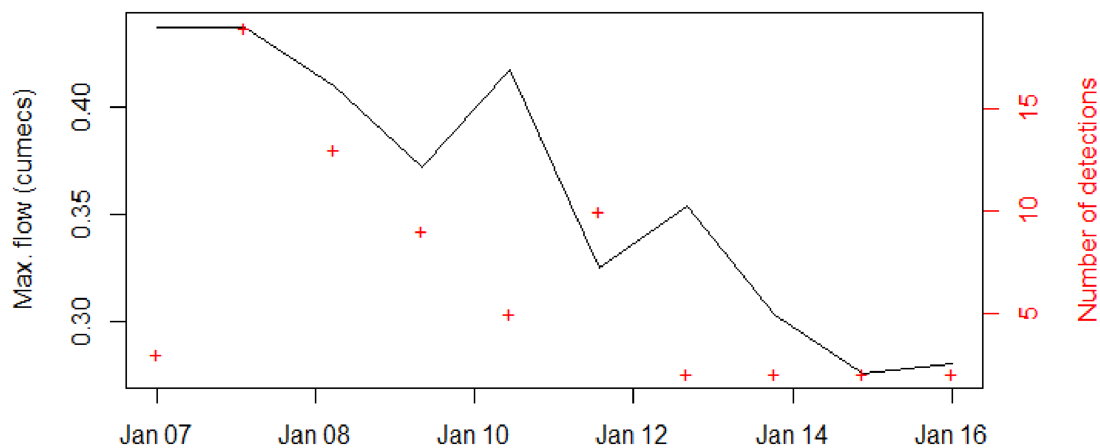

Figure 3.4 Maximum daily flow (m^3) and higher stress antenna detections during year one.

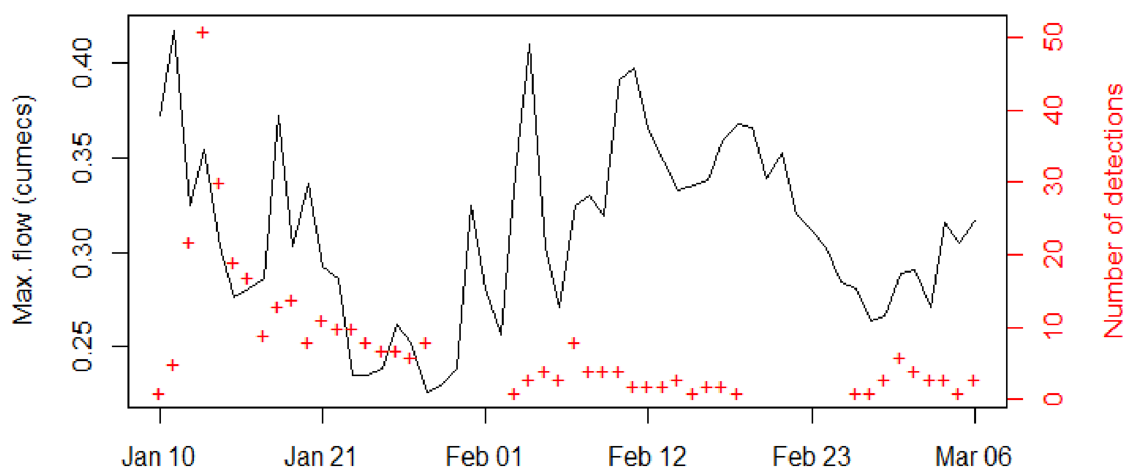

Figure 3.5 Maximum daily flow (m^3) and intermediate stress segment antenna detections during year two.

3.4.6 Fish size and movement

The mean distance moved by 1+ fish (1484 m) in the first survey period was significantly greater than the mean distance moved by 0+ fish (385 m) (*P* =0.01) (Table 3.3). The mean distance moved by 1+ fish was also greater in the second and third study periods, but the difference was not statistically significant (*P*=0.19, *P*=0.06).

Table 3.3 Results of t-test for the null hypothesis that there is no significant difference in the mean distance travelled between 0+ (<130 mm) and 1+ (≥130 mm) fish.

Survey Period	Sample size		Mean distance moved (m)		Df	t-statistic	p value
	Age 1+	Age 0+	Age 1+	Age 0+			
1	15	127	1484	385	1.42	2.82	0.01
2	9	142	900	300	2.3	1.42	0.19
3	7	115	1460	436	6.25	2.3	0.06

3.4.7 Mobile fish

In year one there was a range of upstream and downstream movement patterns displayed by the most mobile fish. Many of the sample fish that displayed considerable movement were 1+ fish (Fish 1-5, 11-13) (Figure 3.7). Some of these 1+ fish (Fish 2-4 and 12) were also observed displaying schooling behaviour (Figure 3.6). In year two some fish released into the higher stress segment displayed considerable upstream movement (Fish 2-5, 7-14 and 16), (Figure 3.8).

Figure 3.6 A school of 1+ trout observed in the intermediate stress segment.

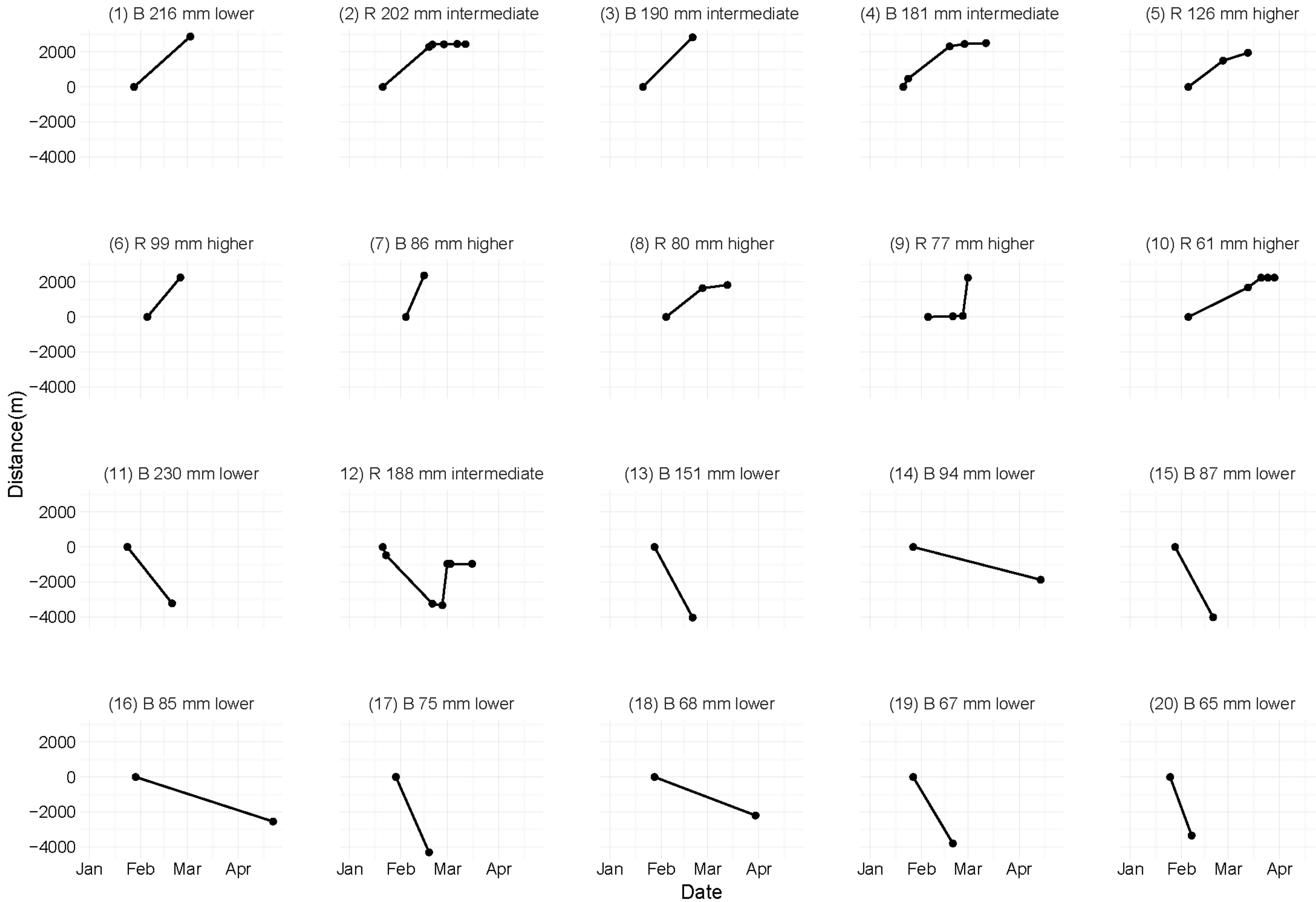

Figure 3.7 Distance moved from release point (0 m) by the 20 most mobile fish in year one. Negative numbers indicate downstream movement, positive numbers upstream movement. The species (brown B, rainbow R), fork length in mm at time of release, and release segment are described for each fish.

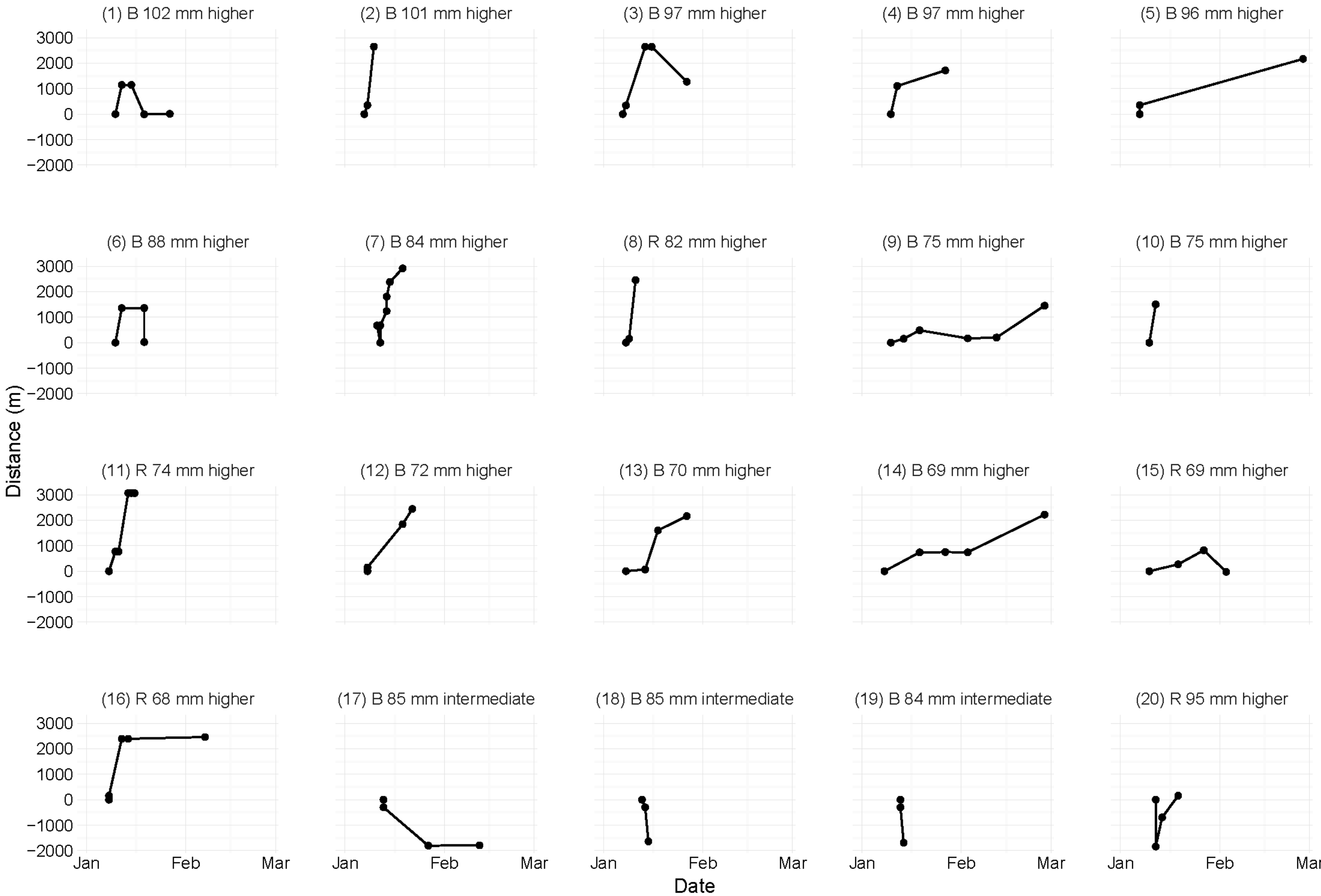

Figure 3.8 Distance moved from release point (0 m) by the 20 most mobile fish in year two. Negative numbers indicate downstream movement, positive numbers upstream movement. The species (brown B, rainbow R), fork length in mm at time of release, and release segment are described for each fish.

3.4.8 Mesohabitat Preference

The majority of fish were detected in riffle habitat (63%), then run habitat (18%) and pool habitat (19 %). The average depth of water that all fish were detected in was 24 cm ±1SD 57 cm (Table 3.4).

Table 3.4 Percentage of sample fish detected in various mesohabitat types.

	Mesohabitat Type			
	Riffle	Run	Pool	Disconnected Pool
Percentage of fish	63 %	18%	7 %	12 %
Mean Depth (cm)	16	22	47	55
±1SD	7	13	18	156

3.5 Discussion

High levels of abstraction resulted in fragmentation of the river corridor preventing fish migration to refuge water such as the Clutha River. To maintain a functional ecosystem for migratory salmonids, longitudinal connectivity is required to allow movement between juvenile rearing streams and adult habitats (Greenberg and Calles 2010). The fragmentation of the river corridor resulted in fish becoming stranded in stressful habitat and was probably a contributing factor to the low survival of the sampled population (see Section 2.4.4).

The observed movement of fish from drying stream segments, and upstream movement patterns (Gowan and Fausch 1996, Kahler et al. 2001) of some fish released into the higher stress segment, was presumably an attempt to leave stressful habitat. However, by the time fish began to move it was not possible for them to access potential refuge water due to the complete drying of riverbed areas. A likely reason for lack of pre-emptive movement during drying events is that fish fail to recognize the disturbance early enough before attempting to move (Davey and Kelly 2007).

There was no indication of large scale downstream movement, which may have signaled an attempted migration event (Holmes et al. 2013), of sample fish past fixed antenna. The most movement recorded past a fixed antenna occurred on January 12 and 13 (year two) at the intermediate stress antenna. The majority of these fish (n = 60) had recently been tagged and released near the antenna (within ~100 m) and may have been repositioning in the river. The other fish (n = 12) detected at the antenna had moved in an upstream direction. The indication of a relationship between flow and fish movements at the higher stress antenna in year two (R^2 = 0.31) may be explained by this antenna dewatering two weeks after deployment. As flows decreased it would have become increasingly difficult for fish to move past the antenna, and this may explain some indication of a relationship between declining flow and fish movement.

Only 1.4 % (n = 7) of sample fish released upstream of the irrigation channel, were later detected by the irrigation channel antenna, and four of these were detected during autumn freshes. This may possibly indicate that fish are more likely to be entrained during higher flows. The presence of a culvert, which can restrict fish movements (Warren and Pardew 1998), as well as shallow areas may have deterred fish from traveling along the channel during low flow conditions.

The considerable movement of some fish along the remaining fragmented river corridor would indicate that had longitudinal passage been possible, some migration to the Clutha River may have occurred. Increased movement of larger 1+ fish, and observations of schooling behaviour which can be associated with migration (Jonsson and Jonsson 2011), may possibly indicate that this year class is more likely to migrate during low flow conditions than 0+ fish. The mean distance moved of 1+ fish during the first survey period of year one was 1484 m, versus 385 m for 0+ fish ($P < 0.05$). It is not known if increased movement by 1+ fish was a natural behaviour, or a response to low flow stress. Elliott (1997) reported that summer low flow conditions can have a more severe impact on 1+ than 0+ cohorts. Jellyman and Bonnett (1992) noted an accumulation of 1+ fish in the lower Lindis River which they thought were trying to migrate to the Clutha River, but these fish also became stranded during low flow conditions. Other studies have documented juvenile trout migration from natal rearing streams to larger waters during summer (Hayes 1988a, Graybill and Palmer 1990, Jellyman and Bonnett 1992, Kristensen 2006).

There was no support for the idea that juvenile trout populations may avoid significant mortality by moving to refuge pools during low flow conditions. The majority of sample fish (63 %) were found within relatively shallow water riffle habitat. As flows reduced during the study period it became more effective to cover pool habitat with mobile antenna (which were submerged to cover all wetted area available) so this result is unlikely to be a detection bias. A possible reason for the preference for riffle habitat during low flows (Davey and Kelly 2007), is that it can provide cover from predators (Brown and Moyle 1991, Henderson and Letcher 2003). Predation was identified as a primary cause of sample fish mortality (see Section 2.5), and can influence habitat choice by prey species (Jackson et al. 2001). Juvenile trout may have sought shallow water habitat to avoid predators such as black shags which were observed targeting fish in pools. Larger sub-adult and adult trout were also observed in many of the pools. The presence of large fish can exclude smaller fish from deep water refuge habitat during drought (Magoulick and Kobza 2003). The broken water and turbulence found in riffle habitat may also provide visual cover (see review in Smith and Brannon 2007) from herons which were seen targeting fish in shallows. Riffle habitat may also allow drift feeding juvenile trout to maximize food (Ayllón et al. 2009), and oxygen intake (see review in Matthews 2012).

There were several limitations of my movement investigation. The fragmentation of the river corridor meant that I was unable to determine if Lindis River juvenile trout migrate out of their natal stream to larger waters during summer low flow conditions. It also restricted the direction (upstream or down) many of the fish could possibly move. There was some indication of a possible relationship between fish movement into the irrigation channel and autumn freshes, experienced following summer low flow conditions, in year one. This relationship was not apparent at mainstem river antenna, possibly because at flows experienced during autumn freshes, fish could pass outside of the proven detection range. Thus, I was also unable to determine if juvenile trout migrated out of the Lindis River during autumn freshes. Another important limitation of my investigation was the very low survival of my sample population (see Section 2.5). This may explain why a large proportion of sample fish were not redetected (451/1016). Fish that were not redetected may have displayed different movement patterns from those that were redetected. However, it is considered very unlikely that a significant number of sample fish left the study area and found refuge.

3.5.1 Management implications and future research

Areas of dry riverbed prevented potential movement to refuge habitat, and further investigation is warranted to understand fish movement response to low flow conditions. Some sample fish displayed considerable movement along the remaining fragmented river corridor, presumably in an attempt to find refuge habitat. Determining how much flow is required to provide fish passage along a river corridor, so that fish experiencing stress may move to refuge habitat is a promising avenue for research and would assist flow management decisions. Juvenile trout movement patterns in heavily abstracted streams warrant further investigation to determine if fish do migrate from natal rearing habitat during summer low flow conditions.

The number of sample fish that entered the irrigation channel during summer low flow conditions was relatively low; however the diversion arm of the irrigation raceway acted as a total barrier to upstream movement of juvenile trout. Provision of fish passage would be warranted to allow access to refuge habitat. Further investigation would be required to determine the rate of entrainment through the autumn season when freshes frequently occur.

Chapter 4: Conclusions and recommendations

High levels of abstraction resulted in the fragmentation of the river corridor and prevented fish access to refuge habitat. If the Lindis River is to be managed to sustain fish populations, it is recommended that the future minimum flow regime provide a longitudinal surface water connection that enables fish movement along the river corridor. A minimal surface connection in drying streams does not ensure fish passage (Magoulick and Kobza 2003) which can be affected by riffle length and predator presence (Schaefer 2001). Determining how much flow and depth is required, so that fish experiencing low flow stress may move along a river corridor to refuge waters warrants further investigation. The provision of fish passage at the irrigation diversion arm is also recommended.

Mark-recapture analysis and subsequent physical habitat monitoring has shown that current levels of abstraction in the Lindis River are resulting in extremely high rates of juvenile trout mortality. Any fish that became stranded in drying pools and died from heat stress were quickly removed by scavengers. This indicates that fish mortalities observed during extended low flow events are likely to represent only a small component of the total mortality. Therefore assumptions that extreme low flow events have limited impact on fish populations due to a lack of observed fish kills (Caruso 2001) are likely to be incorrect. It is expected that high mortality rates reduce the recruitment potential of the Lindis River. Under the present flow management regime the Lindis River is a likely to be acting as a recruitment sink, where the production of spawning runs from larger waters such as the Upper Clutha, and Lake Dunstan is lost

(Gabrielsson 2015). These findings should be taken into account when considering minimum flow options for the Lindis River.

Predation was a much higher cause of juvenile trout mortality than expected. Drought and low flow events can result in severe loss of cover resulting in increased levels of predation (Magoulick and Kobza 2003). Determining the extent of predation in other heavily abstracted streams, and the relationship between available cover and flow reduction, are promising avenues for future research. Results from this investigation indicate that cover from avian predators is particularly important in sustaining juvenile trout populations during low flow conditions. The presence of cover is probably the most important determinant of salmonid populations in streams. Relationships between changes in river flow, predation cover and prey response are very complex (Armstrong et al. 2003). Reliance on habitat simulation models which do not account for these complexities can result in misleading results, and erroneous flow management decisions (Ayllón et al. 2009). It is likely that habitat simulation models which do not consider the relationship between flow reduction and loss of predation cover are overly simplistic.

Mark-recapture techniques combined with stationary cameras offer an opportunity to better understand the relationship between flow reduction, cover, and predation in other stream ecosystems. Several previous studies have experienced difficulty determining a relationship between predation and fish mortality (Boss and Richardson 2002, Berger and Gresswell 2009), and few studies have investigated the relationship between loss of cover and predation during droughts (Magoulick and Kobza 2003). The use of mark-recapture techniques combined with stationary cameras also offers opportunities to quantify the effects of extreme low flow events on native fish. Most native fish species are thought to have higher water temperature tolerances than trout (see review in Caruso 2001), and some may be better able to survive extreme low flow events (Leprieur et al. 2006). However, the observation of large numbers of upland bullies stranded on drying riverbed during my study (Figure 4.1), which quickly disappeared, would indicate that predation and scavenging is also masking the impact of excessive water abstraction on this species. Wild cats and birds targeted this area. Within 24 hours virtually all of the stranded fish had disappeared. The embedded nature of the substrate prevented stranded fish from burrowing. Therefore it is most likely that they were predated and scavenged.

Figure 4.1 Seven (well camouflaged) upland bullies that were still alive and observed under a single cobble within a dewatered reach at day break 6 March 2014.

The protection of freshwater ecosystems from excessive abstraction and flow diversion is a major conservation challenge (Dudgeon et al. 2006), requiring an improved understanding of ecosystem response to flow manipulation (Milner et al. 2012). Excessive levels of abstraction have resulted in degraded aquatic ecosystems throughout many parts of the world (Postel 2000), including arid regions of New Zealand (Lange et al. 2014). Protecting New Zealand river ecosystems from increasing demand for abstraction (Kienzle and Schmidt 2008) is becoming increasingly difficult (Poff et al. 2003). This issue is further compounded by climate change and increased El Nino events, which are likely to result in many dry areas becoming even more arid (see review in Caruso 2002). The development of native tussock grasslands is another compounding factor, which is likely to reduce catchment water yield and stream flow over time (Kienzle and Schmidt 2008, Mark and Dickinson 2008). Mark-recapture techniques as applied in this study have the potential to improve understanding of fish population response to flow reduction. This would enable water managers to make better informed decisions when setting abstraction limits and minimum flows, thereby improving the protection of stream ecosystems.

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